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Evan Cook, Graduate Student

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Environmental and Water Resources Engineering Division
Texas A & M University
College Station, TX 77843

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Despite the current shortage of landfill capacity in densely populated areas of the nation, the relatively low cost of landfill disposal and relatively high human health risk of refuse incineration indicate that landfilling will continue to be the predominant method of refuse disposal.

To control groundwater contamination at municipal landfills, the EPA has proposed new Subtitle D regulations *(Resource Conservation and Recovery Act, 1984)*. Those regulations are premised upon effective leachate control, achieved through utilization of low permeability liners and covers. However, recent research indicates that clay and/or synthetic liners may not prevent leakage of some trace organic compounds in MSW leachate. Furthermore, low permeability covers prevent rapid biological stabilization of refuse, prolonging the liability of landfill owners and operators in addition to delaying return of land to productive use.

This report evaluates the feasibility of alternative landfill technologies including on-site and in-situ biological treatment systems and permeable liner systems. The least expensive and most manageable treatment system would couple in-situ biodegradation with a permeable attenuation layer and secondary aerobic treatment. The attenuation layer would retard the migration of organic compounds and promote precipitation of metallic compounds. On-site biological treatment in aerobic lagoons would remove ammonia and refractory organic compounds.

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ROLE OF LANDFILL DISPOSAL IN MUNICIPAL SOLID WASTE MANAGEMENT

by

Evan Cook

Texas A&M University

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EXECUTIVE SUMMARRY

The current shortage of landfill disposal capacity has prompted the solid waste community to investigate a variety of refuse management alternatives. Available evidence indicates that composting with materials recovery and RDF production with materials recovery are currently the best technologies for refuse pretreatment in rural and urban areas, respectively. However, composting and RDF production only reduce the volume of wastes for final disposal by approximately 45 percent, and landfilling is the only viable option for final disposal. Furthermore, the low cost of direct landfill disposal relative to composting, RDF production, and other alternative disposal technologies indicates that landfilling will continue to be the choice of local decision-makers in many areas of the nation.

To protect human health and the environment, therefore, the solid waste community must identify appropriate landfill technology for municipal refuse management. Recent evidence indicates that clay and/or synthetic liners, which represent the best available technology for municipal landfills, may not prevent leakage of some components of municipal solid waste leachate. In the absence of effective containment, costly and possibly ineffective corrective action would be required under the new Subtitle D regulations. In addition, low permeability covers, which minimize leachate generation, prevent rapid biological stabilization of municipal refuse, prolonging the liability of landfill owners and operators and delaying return of the land to productive use. Thus, leachate treatment is probably a better strategy for groundwater protection at municipal landfills. A permeable attenuation zone, combined with a leachate collection system and secondary leachate treatment might be the least expensive and most manageable leachate treatment alternative.

TABLE OF CONTENTS

	Page
Introduction	1
Cost of Landfill Disposal Versus Resource Recovery	3
Federal and State Regulations	6
Availability of Sites	6
Public Opposition	8
Impacts on Human Health and the Environment	8
Air Contamination	9
Surface Water Contamination	9
Groundwater Contamination	10
Health Risks of Incineration Versus Landfilling	17
Hydrogen Chloride	18
Criteria Pollutants	18
Organic Emissions	20
Metallic Pollutants	22
Municipal Waste Combustion Ash	24
Efficacy of Incineration	27
Efficacy of Recycling	32
Materials Recovery	34
Source Separation	36
Market Demand	37
Disposal Options	41
Ocean Disposal	41
Composting	42
Pyrolysis	44
Anaerobic Digestion	45
Miscellaneous	46

TABLE OF CONTENTS (cont.)

	Page
Future of Landfill Disposal	47
Adequacy of Current Technology	47
Frequency of Leakage	50
Effectiveness of Clay and Synthetic Liners	51
Conclusion	53
Recommendations	54
Future Research	55
References	56

ROLE OF LANDFILL DISPOSAL IN MUNICIPAL SOLID WASTE MANAGEMENT

Introduction

In the 1984 amendments to the Resource Conservation and Recovery Act (RCRA), Congress directed the EPA to evaluate the adequacy of the Subtitle D regulatory program for protection of human health and the environment from groundwater contamination. In addition, Congress specifically ordered the EPA to revise the criteria for facilities that may receive household hazardous waste or small quantity generator hazardous waste (RCRA Section 4010). In response to those directives, the EPA completed a series of studies of Subtitle D waste facilities and Subtitle D waste characteristics.

The research activity at the EPA coincided with several media events, particularly the odyssey of the garbage scow Mobro and the pollution of New York beaches with medical wastes, which focused the public's attention on municipal solid waste (MSW). The manifest problems of the solid waste industry suddenly became a "crisis" in the popular media. That atmosphere intensified the conflict between landfill owners and operators on one side of the solid waste issue and local property owners and environmentalists on the other side.

Opponents of municipal landfills have expressed concerns about pollution caused by landfills, and they have decried the loss of potentially valuable resources. In addition, there has been widespread opposition to landfill development from residents and property owners in the vicinity of proposed landfill sites. These local groups, whose attitudes have collectively been termed the NIMBY (Not In My BackYard) syndrome, have often voiced legitimate concerns about land values, traffic congestion, odors, aesthetics and other problems intrinsically associated with landfill development.

On the other side, landfill owners and operators have defended landfilling as the least costly method of refuse disposal. They have also argued that the data on sanitary landfills did not support assertions by the public that landfills were the principal cause of groundwater pollution (H. Lanier Hickman. January 1988. Waste Age:136). Both sides awaited the verdict of the EPA.

In October 1988, the Subtitle D Report to Congress was published. It concluded:

1) "Shortages of municipal solid waste landfill capacity are occurring in some areas of the nation."

2) "Municipal solid waste landfills have degraded and may continue to degrade the environment."

3) "Human health impacts from exposure to groundwater contamination caused by municipal solid waste landfills have not been documented. However, taken as a whole, the data indicate that releases to the groundwater from municipal solid waste landfills present potential risks to human health."

The findings of the report indicate that the "solid waste crisis" is a real crisis in some areas of the nation where acceptable landfill locations are limited. Furthermore, the findings indicate that available landfill control technology may not provide sufficient protection of groundwater resources.

Currently, approximately 83 percent of the 158 million tons of refuse generated annually in the United States is buried in 6584 active municipal landfills. Eleven percent is recycled, and 6 percent is incinerated. The EPA has estimated that the amount of refuse generated annually will increase 20 percent by the year 2000. However, resource recovery will divert only 18 percent of the refuse generated during the next 15 years from landfill disposal (EPA, Subtitle D Report, 1988).

Although landfill disposal is currently the predominant method of refuse disposal, the number of municipal landfills is decreasing. More than 2000 municipal landfills plan to close within the next five years, yet only 563 new landfills have been opened in the past five years. Furthermore, the number of state approvals for new landfills have dropped almost 25 percent during recent years (EPA, Subtitle D Report, 1988).

The current shortage of landfill capacity has prompted the solid waste community to investigate alternative disposal technologies. The EPA and many solid waste industry analysts advocate a multi-faceted strategy of source reduction, recycling, incineration, and environmentally sound land disposal as an alternative to direct landfill disposal. (EPA, Subtitle D Report, vol. 1, 1988; Eldred 1987) However, refuse disposal decisions are generally made at the local level. Whether direct landfill disposal continues to be the predominant method for solid waste management or a multi-faceted resource recovery strategy is implemented depends largely upon those factors that influence local disposal options: the relative costs of landfilling versus incineration, federal and state regulations, availability of acceptable sites, public opposition, relative risks of landfill disposal versus incineration, the effectiveness of incineration and recycling as waste reduction processes, and the efficacy of other disposal options.

Cost of Landfill Disposal Versus Resource Recovery

The major advantage of direct landfill disposal over other disposal options has been its low cost. According to the 1987 Waste Age tipping fee survey, the average cost of landfill disposal in the Northeast was $39.23/ton. The average cost in the South was $12.27/ton. In the Midwest, it was $12.71/ton, and in the West, $10.75/ton. Nationwide, the average tipping fee increased 51.6 percent from 1986 to 1987. Most of that increase occurred in the Northeast where landfill tipping fees almost doubled (Pettit 1988).

Although the Waste Age survey is the best available indicator of trends in landfill disposal costs; the results are probably not unbiased. Responses were voluntary and presumably do not include the higher spectrum of tipping fees. In the survey, the highest reported tipping fee was $75.00/ton at Fall River, Massachusetts. Substantially higher fees have been reported. In Plainfield, New Jersey, for example, the cost of landfill disposal was $137/ton (Darcey, "States Take Action," 1988). Thus, the true average of landfill tipping fees is probably higher than the averages reported in the survey.

Furthermore, the cost of landfill disposal will increase after promulgation of the new municipal landfill regulations. Under the proposed regulations, the EPA has estimated that the incremental increase in the cost of landfill disposal will be $11.01/ton for landfills with a 10 meter point of compliance (POC) and $9.54/ton for landfills with a 150 meter POC (EPA, Regulatory Impact Analysis, 1988).

The cost of landfill disposal may also increase if the trend toward regionalization continues. Many municipalities in areas where landfill capacity is limited have already opted to haul refuse to regional landfills. In New York, for example, approximately one-third of the refuse generated by the Town of Hempstead (pop. 773,000) is transported 90 miles to Goshen, New York, where it is landfilled. Five states, in responses to the Subtitle D census, specifically cited regional landfills or long distance refuse hauling as solutions to capacity problems in their areas.

Continued opposition to landfill development by NIMBYists will probably accelerate the trend toward regionalization by forcing landfills to be located farther from developed areas. In addition, the economies of scale imposed upon landfills by the proposed new regulations will favor development of regional landfills (EPA, "Notice of Proposed Rulemaking", 1988). Institutional constraints to further development of regional landfills were largely eliminated when the Supreme Court ruled in the case of Philadelphia vs. New Jersey that refuse is entitled to

commerce clause protection. Thus, interstate transportation of municipal refuse can not be be restricted by state legislation (Darcey,"States Take Action," 1988; Logsden 1988). Nevertheless, further development is limited by the cost of transportation. For every additional mile that refuse must be hauled, disposal costs increase by approximately 30 cents per ton (Darcey,"States Take Action," 1988).

Transfer facilities can reduce transporation costs to regional landfills by freeing collection vehicles and their crews from trips to distant disposal sites. Because many different disposal facilities can be utilized without rearranging collection routes, transfer facilities also increase the flexibility of long range planning for solid waste disposal. However, the cost effectiveness of transfer facilities depends upon the distance to disposal sites. Specialized vehicles, equipment, and facilities for transfer stations plus additional labor increase the cost of landfill disposal. In 1975, the cost for specialized transfer equipment and facilities was approximately $.70/ton for a 1000 ton/day facility. For smaller facilities, costs were higher; the cost for a 200 ton/day facility was approximately $1.00/ton (Rimberg 1975).

Additional costs for transfer facilities must be balanced against labor costs for collection crews travelling to distant disposal sites. In 1966, the American Public Works Association estimated that a 110 minute travel time round trip was the break-even point for tranfer facilities. At that time, the cost for hauling refuse that distance was about $2.60/ton (Rimberg 1975). Transportation costs have obviously increased since that time, but labor costs have also increased. Ultimately, the additional expense of hauling refuse to regional landfills makes resource recovery more cost competitive.

In 1987, the average cost of resource recovery nationwide was $33.64/ton, more than ten dollars higher than the nationwide average for landfill disposal (Pettit 1988). In all regions of the nation except the Northeast, the average landfill tipping fees were lower than the nationwide average for resource recovery, even when the estimated compliance costs for the new regulations were added to the landfill fees. Moreover, the cost of resource recovery will probably increase when the proposed air emission regulations for refuse incinerators are enacted.

Those regulations, scheduled for promulgation in September 1991, will require dry alkaline scrubbers and either electrostatic precipitators or fabric filter baghouses for all new and existing municipal waste combustion (MWC) facilities. The EPA has estimated that the annualized cost of the additional air pollution control equipment will be $4 to $9/ton for mass burn facilities and

$4 to $5/ton for facilities that burn refuse derived fuel (EPA 1987). However, a recent study of MWC facilities in Florida, where similar regulations have already been enacted, suggests that the cost of retrofitting existing facilities could exceed the EPA estimate by more than 300 percent (Waste Age,"Plant Retrofits," March 1988).

In addition, the cost of incineration must include expenditures for final disposal of ash and incombustible residues. In mass burn facilities, the volume of ash residues may be 10 to 30 percent of the volume of the incoming refuse (EPA 1987). Refuse derived fuels (RDF) have lower ash percentages, ranging from 10 to 20 percent (Diaz, et al 1982). However, the portion of raw refuse used to produce RDF only comprises 40 to 60 percent of the refuse stream (Diaz, et al 1982). Some of the materials recovered during RDF processing (e.g. ferrous metals and glass) may be recyclable. However, Wolpin (1988) notes that 15 to 20 percent of the refuse delivered to the Reuter, Inc. RDF facility in Eden Prairie, Minnesota went directly to a landfill. In contrast, mass burn facilities usually remove only incombustible items too large for the feed system. Thus, the actual volume reduction achieved through incineration of refuse derived fuels, approximately 50 to 70 percent, may be 20 percent lower than the volume reduction at mass burn facilities. Currently, 90 percent of ash from MWC facilities is buried in municipal landfills or ash monofills (EPA, Subtitle D Report, 1988). However, other methods of disposal, including ocean disposal of ash and incorporation of ash in concrete building blocks, have been investigated (Neal and Schubel 1987).

Lead and cadmium have been detected in ash from municipal incinerators at concentrations in excess of the toxicity limits that define hazardous wastes. If Congress decides to regulate municipal incinerator ash as a hazardous waste, further cost increases for incineration are almost inevitable. Some analysts have predicted price increases of 300 to 400 percent if ash must be processed as hazardous waste (Darcey,"Incinerator Ash Dilemma,"1988).

Currently, the EPA regulates the ash from incinerators accepting only household waste under Subtitle D. In late 1987, however, Congress wrote a letter to the administrator of the EPA that stated, "While the requirements of Subtitle C may create additional costs for municipalities, we are... seriously concerned about the risks to human health and the environment presented by the toxic metals, such as lead and cadmium, that are present in ash in significant quantities." (Darcey, "Incinerator Ash Dilemma," 1988) The EPA is presently evaluating appropriate methods for disposal of municipal incinerator ash (EPA, Subtitle D Report, 1988).

Even if ash is regulated as a nonhazardous waste, however, disposal charges may be higher than regular charges due to the costs of special handling such as dust suppression. For example, Diaz, et al. (1982) noted that additional handling charges for ash disposal in the San Diego area totalled $11/ton.

The low cost of landfill disposal relative to incineration indicates that landfilling will continue to be the predominant method of refuse disposal in areas where acceptable landfill locations are available. Although cost is not the only factor governing the selection of disposal alternatives, it is often the most important factor when projects are publicly funded. Other factors, which may override cost considerations, in the decision process within the public sector include state and federal regulations, availibility of acceptable locations, public opposition, and protection of human health and the environment.

Federal and State Regulations

Federal and state regulations do not seem to be prohibitve in most states. Current federal regulations only restrict landfill development in the vicinity of airports, floodplains and critical habitats of endangered species (40 CFR 257). The proposed regulations for municipal landfills would further restrict development in wetlands, fault areas, seismic impact zones, and karst terrain (EPA,"Notice of Proposed Rulemaking,"1988). In terms of economic impact, the proposed regulations would add about $10/ton to the cost of landfill disposal.

Some state regulations are more stringent. New Jersey, for example, has banned disposal of wastes in landfills that do not employ double liners and leachate collection systems (Darcey,"States Take Action," 1988; Waste Age,"Beach County,"1988). Thirty-six states restrict floodplain development. Thirty-nine states specify minimum distances to man-made structures. Sixteen restrict development in critical habitat. Four have restrictions in geologically sensitive areas, and two have restrictions on soil conditions at landfill sites (EPA, Subtitle D Report, 1988). Only nine states, however, among the forty-six states that responded to the Subtitle D Census, cited permitting requirements or the acquisition of acceptable landfill locations as major reasons for capacity problems.

Availability of Sites

Nevertheless, thirty-one states reported capacity problems. Undoubtedly, some states face a "garbage crisis". However, the reasons for the crisis and its extent are the

subjects of debate. Sheila Prindiville, director of solid wastes disposal for the National Solid Wastes Management Association (NSWMA), has said,"There is no solid waste crisis. We have the land area and the facilities to deal with the nation's solid waste in an environmentally sound and healthful manner."(Darcey 1988, "States Take Action":22) Whether those facilities are available at the right time and place at a reasonable price and whether those facilities are actually environmentally sound and healthful are probably the central issues of the current debate. However, concerning the availability of land, responses to the Subtitle D census seem to confirm the conclusions of the NSWMA. Only five states specifically mentioned the availability of land as a major cause of capacity problems, and only three states cited the cost of land as a major problem.

Thus, the current shortage of landfill capacity may have been caused by lack of planning rather than stringent regulations or scarcity of land. In the Subtitle D Report to Congress, the EPA stated:

> These shortages are becoming critical in densely populated areas of the country, particularly in the Northeast. In addition to limited source reduction and recycling, other factors that appear to be contributing to the capacity problem include difficulty in siting new disposal or treatment facilities due to public concerns and limited long term planning by some state and local governments (EPA, Subtitle D Report, vol.1, 1988:14).

Sixteen of the thirty-one states that reported capacity problems cited only localized problems, which might have been avoided with better planning. Responses from five states clearly indicated that a lack of long term planning had contributed to capacity problems. For example, Maryland, where eight of twenty-four jurisdictions reported less than five years of available landfill capacity, stated, "There is no programmatic mechanism for moving waste from an area with a capacity shortage to an area with a capacity surplus." (EPA, Subtitle D Report, 1988: C-3) Connecticutt's response also indicated a lack of planning. It stated, "No new municipal waste fills have been permitted since 1978. The permitted landfills will be used up before the planned resource recovery projects are in operation."(EPA, Subtitle D Report, 1988: C-2)

Many states, including some states that reported no capacity problems, have started resource recovery programs. Louisiana, for example, reported no capacity problems, but public fears about pollution problems at solid waste landfills have motivated a statewide recycling program

(Darcey,"States Take Action,"1988). In the Subtitle D census, three other states, that reported no capacity problems, cited public opposition to landfills as a major problem.

Public Opposition

In the Subtitle D Census, eleven states reported difficulty siting new landfills due to public opposition. New Jersey, for example, reported, "Capacity problems are very severe across the state. Siting due to public opposition is the largest contributing factor to the capacity problem." (EPA, Subtitle D Report, 1988: C-4) In contrast, Arkansas stated," ...no problems are foreseen in finding new locations. This is primarily due to a 1974 Arkansas ruling which said that landfills can only be turned down because of physical criteria siting problems, not because of public opposition." (EPA, Subtitle D Report, 1988: C-1)

Residents and property owners in the vicinity of proposed landfill sites often have legitimate concerns about the impact of landfill activities on the quality of life and value of property in their locale. Many people in these local groups also share a widespread concern about pollution caused by landfills. The public's fear of landfills may be due to confusion between hazardous and municipal waste landfills, stemming from pre-RCRA disposal of hazardous wastes in municipal landfills. Nevertheless, their fears are not completely unjustified.

Impacts on Human Health and the Environment

The Subtitle D Report to Congress cited 845 air contamination violations, 586 groundwater violations, and 660 surface water violations at municipal landfills during 1984. These violations suggest that the current level of technology employed at municipal landfills is insufficient protection against environmental contamination. However, due to differences in monitoring programs at landfills, these data are only a gross measure of the impacts of municipal landfills on human health and the environment (EPA, Subtitle D Report, 1988).

Violations are defined differently among the states; some facilities may have been cited for more than one violation, and some violations may have been overlooked by lax inspection programs. Furthermore, monitoring at municipal landfills has generally been lax. Only 9 percent of municipal landfills monitored landfill gas, and only 3 percent monitored air. Less than 50 percent monitored groundwater. Only 15 percent monitored surface water (EPA, Subtitle D Report, 1988).

Air Contamination

Migration of landfill gas from municipal landfills presents substantial human health risks. The composition of landfill gas varies during the landfill stabilization cycle; however, it generally consists of 50 to 60 percent methane and 40 to 50 percent carbon dioxide with trace quantities of other gases including volatile organics. Emissions of volatile organic compounds have been estimated to be 200,000 megagrams per year. Benzene, toluene, tetrachloroethylene, vinyl chloride, and xylene have been detected in landfill gas at concentrations exceeding OSHA limits. (EPA, Subtitle D Report, 1988).

Twenty-nine damage incidents involving landfill gas have been documented by the EPA. Twenty-one of the incidents involved fire or explosion; fifteen of the incidents caused injuries, and five caused fatalities. Eight people, in total, have died due to inadequate control of methane, generated by municipal landfills. In addition, high levels of vinyl chloride were detected during two damage incidents, and high concentrations of benzene were detected once (EPA, Subtitle D Report, 1988).

In the new municipal landfill regulations, the EPA has proposed more stringent requirements for landfill gas control. The Criteria for Classification of Solid Waste Disposal Facilities and Practices (40 CFR 257), promulgated in 1979, limited the concentration of methane in facility structures to 25 percent of its lower explosive limit. Furthermore, it limited the concentration of methane at facility boundaries to the lower explosive limit. Apparently, these criteria were inadequate because twenty-one of the twenty-nine incidents cited above occurred after 1979. Therefore, the new regulations require periodic monitoring for methane, and corrective action when the promulgated limits are exceeded (EPA,"Notice of Proposed Rulemaking," 1988). The EPA is currently studying appropriate standards for emissions of volatile organic compounds (EPA, Subtitle D Report, 1988).

Surface Water Contamination

No incidents have been reported of human health or economic impact caused by contamination of surface water resources. The absence of such incidents may be attributed to the lack of landfills near surface water bodies. Fewer than 100 landfills are located within one mile of a surface water body used for drinking water.

Although human health impacts have not been documented, surface water contamination at municipal landfills has probably caused substantial environmental damage. Seventy-three incidents of surface water degradation have

been documented by the EPA, including thirteen incidents of ecological damage (EPA, Subtitle D Report, 1988). However, the number of landfills where surface water has been polluted is probably much greater than the number of documented incidents. The EPA case studies indicate that the major factors contributing to surface water contamination at municipal landfills are leachate seeps, run-off control deficiencies, and floodplain or wetland locations. The municipal solid waste landfill survey data indicate that approximately 2600 active municipal landfills were designed without surface water run-on/run-off control systems; 766 were located within a 100-year floodplain, and 334 were located in wetlands (EPA, Subtitle D Report, 1988). Thus, many existing landfills may be prone to surface water pollution problems.

Although the ecological impact of leachate contamination depends upon the characteristics of the leachate and the attenuation capacity of the receiving body of water, the potential for environmental damage caused by landfill runoff is substantial. Cameron (1978) reported that 7 gallons of a "typical" leachate diluted in 100 gallons of water killed 50 percent of the fish in a bioassay within 96 hrs. Six gallons of a "concentrated" leachate diluted in 10,000 gallons of water killed 50 percent of the test organisms within 96 hours. However, ecological damage may also be subtle, and therefore, it may not be reported under normal circumstances. For example, the results of one five year study of a stream near a municipal landfill indicated that the diversity of benthic organisms downstream from the landfill was substantially less than the diversity upstream. (EPA, Subtitle D Report, 1988) Given that only 15 percent of municipal landfills monitor surface water, such impacts would probably not attract much attention. Since forty percent of municipal landfills are located within a quarter mile of a surface water body, ecological damage caused by landfill runoff may be wide-spread.

The proposed new regulations would provide greater protection of surface water resources by including requirements for run-on/run-off control systems. Section 257.3-3 of the 1979 solid waste disposal criteria prohibited the discharge of pollutants in violation of the Clean Water Act. However, the EPA did not have authority to enforce that prohibition. Under the 1984 amendments of RCRA, the EPA now has enforcement authority.

Groundwater Contamination

The EPA has documented degradation of groundwater resources at 146 municipal landfills. Drinking water resources have been adversely affected at 35 sites. At 17 sites, alternative water supplies have been required (EPA, Subtitle D Report, 1988). The historical disparity between

incidents of groundwater contamination and well contamination is probably due to the scarcity of wells in the vicinity of municipal landfills rather than the effectiveness of containment technology. Currently, 54 percent of municipal landfills have no downgradient drinking water wells within one mile (EPA, Subtitle D Report, 1988). Historical evidence indicates, however, that development will encroach upon municipal landfills. For example, Garland and Mosher (1975) cited an incident of groundwater contamination from a dump, once considered to be remote, which threatend wells in a subdivision, necessiataing an off-site water supply at a capital cost of $500,000 and annual costs of $55,000. Current state regulations prescribe distances from municipal landfills to community water supplies ranging from 400 feet to one mile (EPA, Subtitle D Report, 1988), yet leachate plumes have been detected 5000 to 10,000 feet from municipal landfills (Shuster, Sayville Case Study, 1976; Kimmel and Braids 1980; Palmquist and Sendlein 1975).

Several incidents of well contamination have caused substantial human hardship and economic losses. For example, In Rockford, Illinois, four residential wells, four industrial wells, and one public water supply well were abandoned when leachate from the Peoples Avenue landfill contaminated a water supply aquifer. The cost to develop alternative water supplies was over $200,000 (Shuster, Peoples Avenue Case Study, 1976). In Aurora, Illinois, seven residential wells were contaminated by leachate from the Fox Valley landfill. The cost for an alternative water supply was $115,000 (Shuster, Fox Valley Case Study. Leachate from the Sayville landfill contaminated three residential wells near Islip, New York (Shuster, Sayville Case Study, 1976), and in Genesco, Illinois, Chian and DeWalle (1976) reported contamination of the municipal water supply by leachate from a garbage dump.

In New Castle County, Delaware, leachate from the Army Creek (Llangollen) landfill contaminated a residential well and threatened the most productive well field in the state. To prevent further contamination of the aquifer, groundwater recovery wells and a monitoring network were installed at a cost of 1.35 million dollars. Estimates of the final remediation costs ranged from more than one million dollars for capping the landfill, installing leachate pumps, and treating the leachate at a sewage treatment plant to fifteen million dollars for complete removal of the refuse (Apgar 1975; DeWalle and Chian 1979).

Health risks from exposure to groundwater contaminated by MSW leachate are not known. No incidents of illness or death caused by MSW leachate have been reported. Thus, no epidimeological data are available. Furthermore, the composition of municipal leachate is highly variable and site-dependent. Factors that affect leachate composition

include characteristics of the refuse, the age of the landfill, hydrogeologic and climatic conditions at the landfill, and the design and operation of the landfill. Because the composition of leachate is variable, the potential for human health impacts from leachate exposure is also variable and assessment of potential risks is difficult.

Nonetheless, the toxicities of trace organic and inorganic constituents of municipal leachates have generated some concern. Brown and Donnelly (1988) compared the carcinogenic potencies of two municipal landfills to the carcinogenicity of the Love Canal landfill. Fifty-one Appendix IX (RCRA) organic pollutants and thirteen heavy metals have been identified in municipal landfill leachates. The median concentrations of 19 of the 51 Appendix IX organics exceeded EPA drinking water or health based standards. Maximum concentrations of seven additional organics exceeded prescribed limits. Median concentrations of cadmium, chromium, copper, lead, manganese, and zinc were greater than prescribed standards, and maximum concentrations of iron, mercury, and nickel also exceeded water quality criteria or health based limits.

In addition, median concentrations of antimony and thallium exceeded EPA limits for systemic toxicants, and the median concentration of selenium was greater than the maximum contaminant level of the Safe Drinking Water Act. Arsenic and barium have been detected at concentrations in excess of the maximum contaminant levels, and nitrate has been detected at concentrations greater than the EPA's water quality citeria (EPA, Subtitle D Report, 1988).

The data base for concentrations of trace organic and inorganic constituents includes leachate analyses from landfills constructed prior to the 1980 RCRA prohibition on disposal of hazardous wastes in municipal landfills. Therefore, the results may not be representative of landfills that accept only municipal refuse. However, the EPA found no significant differences between pre-1980 and post-1980 leachate characteristics (EPA, Subtitle D Report, 1988).

In situ soils beneath landfills have some capacity to attenuate pollutants. Therefore, the risk associated with consumption of raw leachate is not a good measure of the actual human health risk posed by leachate contaminated groundwater. Ham (1975) noted, "There are several case studies in the literature in which the contamination of groundwater was shown to drop off considerably with distance from the landfill and, for most soil and water flow conditions, this appears to be the norm."

The EPA has estimated the carcinogenic risk potential from consumption of wellwater contaminated by municipal landfills. Their modelling results indicate that 17 percent of the 6000 existing municipal landfills in the United States pose a carcinogenic risk greater than 1:1,000,000 (EPA, Subtitle D Report, 1988). However, there are several important caveats to the EPA modelling data.

The groundwater model assigns zero risk to landfills with no wells within a one mile radius. Since leachate plumes have been detected 10,000 feet from municipal landfills (Kimmel and Braids 1980), this assumption is questionable. The model also does not consider the construction of new wells within a one mile radius of existing landfills or the construction of new landfills within a one mile radius of existing water supplies (EPA, Subtitle D Report, 1988). Both scenarios are possible under current regulations in some states. Because 54 percent of the landfills in the model are assigned zero risk, the model probably underestimates the actual risk from leachate contamination. When risk was calculated for the subset of landfills with wells within one mile, 40 percent of the facilities exceeded the 1:1,000,000 cancer risk (EPA, Subtitle D Report, 1988).

The model also assumes that the size distribution for landfills will not change. However, current trends in the solid waste industry indicate that the average capacity of municipal landfills will increase. Since larger facilities are generally associated with higher levels of contamination (EPA, Subtitle D Report, 1988), this assumption may also cause the model to underestimate the human health risk from municipal landfills.

Finally, the model calculates risk for only eight leachate constituents. Although the eight constituents were chosen for their pollution potential, mobility, and persistence in the subsurface environment, the human health risk of these eight constituents can not be extrapolated to predict the risk posed by the gamut of potential pollutants in MSW leachate. Moreover, the model does not consider synergistic or antagonistic interactions between leachate constituents. The Subtitle D Report states, "The model components introduce considerable uncertainty, particularly those that predict leachate quality for trace organics ... and the human health risk resulting from exposure to toxic substances."(EPA, Subtitle D Report, vol.1, 1988:23)

Only comprehensive field data on the movement and attenuation of MSW leachate plumes could provide an unequivocal risk determination. Unfortunately, such data is limited, particularly for trace organics. Only about 40 percent of the pre-1980 and 50 percent of the post-1980 landfills monitored groundwater (EPA, Subtitle D Report, 1988). Moreover, the monitoring techniques employed at

municipal landfills may have been inadequate for detection of leachate contamination. Several investigators have reported that MSW leachate plumes sink to the bottom of contaminated aquifers due to density and viscosity differences between leachate and uncontaminated groundwater (Cherry 1981, Kimmel and Braids 1980). Consequently, multi-level sampling is necessary to detect leachate migration. In addition, Kimmel and Braids (1980) suggested that leachate may migrate as slugs of contaminated groundwater after rainfall events. Thus, periodic monitoring may fail to detect leakage.

Furthermore, routine monitoring for indicator parameters may not detect trace organic contamination. Reinhard, et al. (1984) found that dense non-aqueous phase liquids in MSW leachate plumes, particularly chlorinated solvents, tend to sink below chloride in leachate plumes, entraining other organics with densities less than water. Robertson, et al (1974) suggested that low level contamination by hazardous organics would not cause detectable alterations of polluted water. Thus, routine monitoring for chloride, conductivity, total dissolved solids, etc. may be inadequate for detection of trace organics.

Given the inadequacy of past monitoring practices, the available field data can not be used to support a conclusion, negative or positive, on the risk posed by MSW leachates, particularly the trace organic components. Nevertheless, the available data provides sufficient reason for concern. Chronic ingestion of even low concentrations of some of the compounds identified in municipal leachate plumes could constitute a substantial human health risk. For example, the 1:1,000,000 cancer risk for benzene is 0.67 micrograms per liter (Wilson, et al 1986). Reinhard, et al. (1984) reported detection of benzene at 2.1 micrograms per liter 800 meters downgradient from the North Bay landfill near Ontario. Chian and DeWalle (1979) reported a benzene concentration of 4.3 micrograms per liter 500 meters from the Army Creek landfill in Delaware. Chian and DeWalle also detected acetone, cloroform, dichloroethane, trichloroethylene, benzene, tetrachloroethylene, and toluene. Both dichloroethane and chloroform were detected at concentrations exceeding the 1:1,000,000 cancer risk. The Subtitle D Risk Model identified halogenated aliphatic hydrocarbons as the principal human health risks among the eight leachate constituents modelled by the program (EPA, Subtitle D Report, 1988).

Small quantity generator wastes and discarded containers of household hazardous wastes have generally been considered the primary sources of trace organic contaminants in municipal solid wastes. However, Robertson, et al (1974), Cameron (1978), and Tester and Harker (1982) have speculated that some trace organics may also originate from the decomposition of domestic products, such as garden hoses,

electrical insulation, and upholstery. If their suppositions are correct, then municipal landfills may function as reservoirs that release trace levels of complex organic chemicals into the environment for years after closure of the landfill (Robertson, et al 1974).

The concentrations of heavy metals in leachate plumes have generally been moderate, iron and manganese being notable exceptions. (Kimmel and Braids 1980, Cherry 1981). Heavy metals in refuse primarily occur in their metallic states. Consequently, their solubilities in water are low, and only trace quantities are leached from refuse (Kimmel and Braids 1980). Additionally, heavy metals in MSW leachates precipitate as metallic salts, particularly metallic sulfides, due to low redox conditions prevalent in municipal leachates (Chian and DeWalle 1976, Kimmel and Braids 1980, Cherry 1981). Metals may also be removed by cation exchange and/or adsorption onto soil colloids.

Griffin and Shimp (1978) found that precipitation was the principal attenuation mechanism for lead, cadmium, mercury, and zinc. Other heavy metal cations, including chromium (III) and copper, were attenuated primarily by an adsorption-exchange mechanism. Attenuation of iron and manganese was controlled by redox potential. Under low redox conditions, iron and manganese were unattenuated. Moreover, under such conditions, iron and manganese oxide coatings were eluted in substantial concentrations from clay and sand particles. Under less anaerobic conditions, substantial attenuation occurred (Griffin and Shimp 1978).

Iron has been the most prevalent metal detected in municipal leachates. It was detected in 55 of 62 analyses of MSW leachates compiled by the EPA (EPA, Subtitle D Report, 1988). Iron has also been detected at substantial concentrations in leachate plumes from municipal landfills. Kimmel and Braids (1980) detected iron 3000 feet from the Babylon landfill in New York at concentrations ranging from 320 to 690 mg/l. Cherry (1981) detected iron at 300 mg/l in a leachate plume from the North Bay landfill near Ontario. Both research groups noted, however, that iron may have been contributed by reductive dissolution of iron oxide coatings from aquifer materials. Generally, concentrations of iron in MSW leachates have been lower than the EPA water quality criteria (1000 mg/l). The median concentration of iron from 55 leachate analyses was only 95 mg/l.

In contrast, the median concentration of manganese (3.7 mg/l) was 74 times higher than the water quality criteria (0.05 mg/l), but it was detected in only 43 leachate analyses (EPA, Subtitle D Report, 1988). Kimmel and Braids (1981) detected manganese 3000 feet from the Babylon landfill at concentrations ranging from 80 to 600 mg/l.

Cherry detected it in the leachate plume from the North Bay landfill at 21 mg/l. However, he suspected that manganese, like iron, might have eluted from aquifer materials.

Kimmel and Braids (1980) also detected arsenic and selenium in leachate plumes from the Babylon and Islip landfills. Arsenic was detected at 36 of the 62 landfills where leachate analyses were conducted. However, its median concentration (0.0135 mg/l) in MSW leachates is substantially lower than the maximum contaminant level (0.05 mg/l). In contrast, the median concntration of selenium (0.02 mg/l) is two times higher than the maximum contaminant level (0.01 mg/l), but it was detected at only 17 landfills (EPA, Subtitle D Report, 1988).

Kimmel and Braids (1980) reported that arsenic and selenium were present in the leachate plumes from the Babylon and Islip landfills as arsenate and selenate. They noted that these anions were not strongly adsorbed on materials in the aquifers. Their findings substantiate the laboratory results of Griffin and Shimp (1978) who found that the principal removal mechanism for anionic compounds, including arsenic, selenium, and chromium (VI), was adsorption of the monovalent species. In general, Griffin and Shimp concluded that the removal efficiencies of anionic compounds were substantially lower than the removal efficiencies for cationic compounds.

Griffin and Shimp (1978) concluded that calcium-saturated clay soils generally provided good removal of heavy metals, except iron and manganese under low redox conditions. However, the effectiveness of clay liners for attenuation of heavy metals would depend upon the chemical composition of the leachate, the pH and redox potential of the leachate, the cation exchange capacity of the clay materials, the type of cations initially present on the exchange complex, and the condition of the liner. Adsorption of heavy metals from MSW leachate was 50 to 90 percent lower than adsorption from uncontaminated water.

The 1979 Criteria for Solid Waste Disposal Facilities and Practices contained provisions which required that pollutants in leachate contaminated groundwater not exceed the maximum contaminant levels of the Safe Drinking Water Act. The new regulations would further specify groundwater monitoring requirements and mandate corrective action if groundwater protection standards set by the states were exceeded. Nevertheless, further incidents of groundwater contamination and costly remedial cleanups are probable unless better technologies for groundwater protection are developed

Although the data on leachate migration and attenuation are limited, available information indicates that groundwater contamination by MSW leachates may pose

substantial human health risks. Furthermore, substantial theoretical evidence indicates that the best available landfill technology may not prevent the leakage of some trace organic compounds. The absence of human health impacts to date is probably due to the small number of public water supply wells in the vicinity of municipal landfills rather than the effectiveness of current landfill technologies.

The new municipal landfill regulations may partially assuage the public's fear of landfills; however, continued opposition to landfill disposal is probable. The effect of that opposition on development of new landfill disposal capacity is, however, difficult to assess. One variable is the price that taxpayer-consumers will pay to forego threats of pollution from landfills. Another variable is the impact of public opposition toward incineration, the most likely alternative to landfills, on the implementation of a resource recovery strategy. Like opposition to landfills, opposition to incineration is often generated by concerns about the quality of life and value of property in the vicinity of combustion facilities. However, it may also reflect concerns about human health risks.

Health Risks of Incineration Versus Landfilling

A comparison of the projected cancer risks associated with incineration and landfilling suggests that incineration presents a greater human health risk than landfill disposal. The estimated cancer risk from direct inhalation of incinerator stack emissions is 2 to 20 deaths per year nationwide. Furthermore, incineration poses a cancer risk, comparable to the direct exposure risk, from indirect exposure to persistent organic compounds, such as chlorobenzenes, PCBs and dioxins, which accumulate over time in the soils surrounding combustion facilities (EPA 1987). In comparison, the cancer risk from exposure to groundwater contaminated by MSW leachate has been estimated by the EPA to be 0.0770 cancer deaths per year nationwide under current conditions (EPA, Subtitle D Report, 1988).

Numerous caveats are associated with the two models used to predict these risks. Probably, the groundwater model underestimates the true risk from leachate contamination; whereas, the air quality model may overestimate the actual human health risk from incinerator emissions. Nevertheless, the magnitude of the difference between these predictions suggests that incineration poses a greater cancer risk than landfilling.

Caveats associated with the incineration model include uncertainties about potential interactions between constituents of incinerator emissions, toxicities of

specific species of polychlorinated dioxins and dibenzofurans, estimation of exposure levels, and extrapolation of carcinogenic potencies of chemical compounds from animal models to humans (EPA 1987).

Potentially harmful emissions from municipal incinerators include hydrogen chloride, carbon monoxide, nitrogen oxides, sulfur oxides, polychlorinated biphenyls, chlorinated aromatics, polycyclic aromatic hydrocarbons, polychlorinated dioxins and dibenzofurans, and heavy metals.

Hydrogen Chloride

In a 1987 study, the EPA predicted that approximately 220,000 tons of hydrogen chloride would be emitted annually from MWC facilities being planned or built or operational at the time of the study (EPA 1987). Although hydrogen chloride is the preferred end product for chlorine molecules released during refuse combustion, it may contribute to local air pollution problems.

Incomplete combustion of chlorine compounds in municipal refuse threatens releases of toxic organochlorine molecules. Under conditions of complete combustion, however, chlorine molecules combine with hydrogen to form hydrogen chloride, which can be removed with acid gas scrubbers (Neal and Schubel 1987). Pilot-scale experimental studies, conducted under the auspices of Environment Canada, indicated that MWC facilities equipped with acid gas scrubbers and fabric filters could remove 90 percent of the hydrogen chloride generated during refuse incineration (EPA 1987).

Adverse health effects from exposure to hydrogen chloride in the vicinity of combustion facilities are not anticipated, but ambient levels may be sufficiently high to cause corrosion of ferrous metals (3 micrograms Cl/cubic meter). Modelling efforts by the EPA indicate that the majority of existing municipal waste combustion facilities exceed allowable levels for hydrogen chloride, based upon materials damage limits. Those models also predict that large mass burn and RDF facilities and all modular combustion facilities currently being planned or built could exceed corrosion limits for hydrogen chloride (EPA 1987).

Criteria Pollutants

Criteria pollutants, as defined by the National Ambient Air Quality Standards of the Clean Air Act, include carbon monoxide, nitrogen oxides, and sulfur oxides (EPA 1987).

In 1975, nationwide carbon monoxide emissions totaled 96.2 million tons; solid waste disposal generated approximately 3.3 million tons (Salvato 1982). Since the

1970's, however, significant improvements in grate design and combustion air control have been achieved. Those improvements have occasioned more complete combustion of carbon compounds. Under conditions of complete combustion, carbon compounds are converted to carbon dioxide. In a well-designed, well-operated, modern refuse incinerator, emissions of carbon monoxide, an incomplete combustion product, should not exceed 100 parts per million (Neal and Schubel 1987).

Given these recent improvements, the EPA has estimated that nationwide emissions of carbon monoxide from refuse incinerators will be approximately 93,000 tons annually (EPA 1987). Although these emissions may be significant locally, on a nationwide scale, their magnitude is small, particularly in comparison to automobiles, which generate approximately 70 million tons of carbon monoxide annually. (Neal and Schubel 1987).

Nitrogen oxide emissions from solid waste disposal facilities totaled 200,000 tons nationwide in 1975 (Salvato 1982). In 1987, the EPA estimated that nitrogen oxide emissions from refuse incinerators would be approximately 148,000 tons annually (EPA 1987). Thus, little improvement has been evident despite recent improvements in incinerator design.

Control of nitrogen oxide emissions is generally more difficult than control of other criteria pollutants. Existing control methods are ill-defined and potentially expensive. Operational control methods include flue gas recirculation. burner modifications, and reduced air preheat. Catalytic denitrification has also been attempted in some facilities. However, poisoning of the catalyst by constitiuents of flue gas is a potential problem in such systems (Diaz, et al. 1982).

Data from a 1979 study of a modular incineration facility in North Little Rock, Arkansas indicated that emissions of nitrogen oxides would exceed the 225 ppm limit of the San Diego Air Pollution Control District. Although high nitrogen oxide emissions from the North Little Rock facility may have been caused by poor combustion controls during the tests, these data indicate that nitrogen oxide emissions from refuse incinerators could cause air pollution problems locally (Diaz, et al 1982).

On a nationwide scale, however, emissions from natural sources, automobiles and other sources overshadow emissions from MWC facilities. Studies in Florida, which boasts the greatest number of existing MWC facilities, found that solid waste facilities contributed less than one percent of the state's nitrogen oxide burden (Waste Age,"Plant Retrofits",1988).

The EPA has estimated that sulfur oxide emissions from MWC facilities will be approximately 70,000 tons annually (EPA 1987). Like nitrogen oxide emissions, however, the magnitude of these emissions is overshadowed by emissions of sulfur oxides from other sources. The low sulfur content of refuse generally precludes sulfur oxide emission problems (Diaz, et al. 1982). In Florida, less than one percent of sulfur dioxide emissions were attributed to solid waste disposal facilities (Waste Age,"Plant Retrofits", 1987). Moreover, acid gas scrubbers and fabric filters can remove 70 percent of sulfur dioxide emissions generated by refuse combustion (EPA 1987).

Thus, emissions of criteria pollutants probably will not present substantial human health risks if acid gas scrubbers and effective particle removal equipment are employed. Emissions of hydrogen chloride may cause materials damage problems but probably will not impact human health. Nonetheless, emissions of these pollutants from new MWC facilities must be critically evaluated in terms of their contribution to the local air pollution burden. In addition, foresight may warrant concern about emissions of carbon dioxide from refuse incinerators. Since carbon dioxide is a "greenhouse" gas, direct conversion of organic wastes to carbon dioxide, outside the realm of natural biological systems, may soon be considered by the general public to be an unwarranted disruption of the natural balance of the biosphere.

Organic Emissions

According to EPA modelling studies, the greatest potential human health risks from refuse incineration are associated with emissions of complex organic compounds, particularly dioxins and furans. Emissions of dioxins and furans were credited with 0.8 to 20 of the total projection of 2 to 20 cancer deaths per year for refuse incineration (EPA 1987). Other toxic organic chemicals (chlorobenzenes, chlorophenols, formaldahyde, polycyclic aromatic hydrocarbons, and polyclorinated biphenyls) were credited with 0.2 to 3.0 deaths per year.

Concentrations of potentially harmful organic compounds may be reduced substantially during incineration if each element of refuse and its combustion products are heated to temperatures greater than 1800 degrees fahrenheit for a minimum of one second. Normal incinerator temperatures range between 1650 and 1850 degrees fahrenheit; however, temperature control is difficult due to the heterogeneous nature of municipal refuse (Neal and Schubel 1987, Reason 1986). Good grate design and a skillful crane operator, who can mix refuse to obtain a uniform heating value, are essential for complete combustion and effective control of organic pollutants.

Substantial improvements have been made in grate design and combustion air control during recent years. Nonetheless, the EPA predicts annual emissions of approximately 2500 kilograms of chlorobenzenes, 3500 kilograms of chlorophenols, 21 kilograms of chlorinated dioxins and dibenzofurans, 119,000 kilograms of formaldahyde, 21,000 kilograms of polychlorinated biphenyls, and 4,000 kilograms of polycyclic aromatic hydrocarbons nationwide (EPA 1987).

Acid gas scrubbers and fabric filters or electrostatic precipitators can be used to control such emissions. Condensation with subsequent removal as particulates or aerosols has been postulated as the primary mechanism of organic pollutant capture (EPA 1987). Thus, post-combustion conditions are important for effective removal. Flue gas temperatures must be lowered substantially from combustion chamber temperatures. For optimum removal of chlorinated aromatics and polychlorinated biphenyls, flue gas temperatures should range between 125 and 140 degrees centigrade. Temperatures should range between 110 and 140 degrees centigrade for optimum removal of dioxins and furans. Given those conditions, results from the Environment Canada studies indicated 99 percent removal of dioxins and furans and 95 percent removal of chlorinated aromatics and polychlorinated biphenyls were possible with a combination of acid gas scrubbers and fabric filters (EPA 1987). However, normal flue gas temperatures may be substantially higher than the temperatures in the Environment Canada tests. In 1978, for example, Greenburg, et al. reported flue gas temperatures of 210 and 260 degrees centigrade at two municipal incinerators in the Washington D.C. area.

Some scientists argue that decomposition will prevent toxic accumulations of complex organic compounds. According to Penner and Weisenhahn, for example, the half-life for photochemical decomposition of 2,3,7,8-TCDD is less than one year in soil and only hours in air (Neal and Schubel 1987). However, results of epidemiological studies in Europe apparently refute those arguments. Those results indicated that human populations were exposed to dioxins from municipal waste combustion facilities, and furthermore, some dioxin compounds accumulated in the milk of nursing mothers (Pollock 1987).

Human health impacts from exposure to dioxins and furans are difficult to assess due to uncertainties associated with predictions of human toxicities and exposure levels. Some species of dioxins and furans are extremely toxic to certain species of laboratory animals. In addition to being carcinogenic, dioxins have been proven to affect fetal development and reproductive systems in small laboratory animals. Some dioxins, such as tetrachlorodibenzo-p-dioxin, may also weaken immune systems, thereby increasing susceptibility to carcinogens. However,

lethal dosages vary among species. For example, the 50 percent lethal dosage for guinea pigs is 1 microgram of dioxin per kilogram of body weight; whereas, the lethal dosage for hamsters is 5,000 micrograms per kilogram. Human toxicities are apparently relatively low. Exposure during industrial accidents to 2,3,7,8 tetrachloro-dibenzo-p-dioxin (TCDD), the most toxic dioxin known, caused chloracne and short-term liver damage. However, no carcinogenic effects have been identified to date (Neal and Schubel 1987).

In 1986, Sweden, which incinerates half of its municipal refuse, became the first nation to regulate dioxin emissions (Pollock 1987). Denmark, the Netherlands, and Ontario also have guidelines for emissions of dioxins and furans, and the EPA is developing emissions criteria for those compounds. Currently, the EPA complies unofficially with the guidelines of the Hernandez document, which sets emissions criteria 20 times more stringent than Ontario's guidelines and 1000 times more stringent than the Netherland's guidelines. Although these emissions standards are relatively strict, available data indicates that a well designed and well operated municipal waste combustion facility should be able to meet those guidelines (Neal and Schubel 1987).

Metallic Pollutants

Cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, zinc, sodium, magnesium, potassium, selenium, arsenic, antimony, barium, and silver have been detected in fly ash and suspended particulates from MWC facilities. Refuse combustion may be the major source of zinc, cadmium, and antimony in many cities. Silver and tin may also originate primarily from refuse incinerators in some areas (Greenburg, et al. 1978). As lead is phased out of gasoline, MWC facilities may become major sources of that element (Campbell 1976). In addition, mercury is probably released in substantial quantities; however, due to its high vapor pressure, approximately half of the mercury in stack emissions remains in the vapor phase at temperatures characteristic of flue gases (EPA 1987, Neal and Schubel 1987).

Among those elements generated in substantial quantities by MWC facilities, cadmium, tin, antimony, lead, and mercury are known to be toxic. Detrimental health effects are enhanced because these metals are predominantly associated with small, respirable particles less than 2 micrometers in diameter. In addition, these materials may be soluble in water and dilute acids. Thus, the particles may be deposited deep in the lungs where the toxic elements are dissolved in bodily fluids and mobilized (Greenburg, et al. 1978).

Enrichment of toxic metals on particulates is probably caused by vaporization of volatile metals and metallic compounds in the combustion chamber, followed by condensation of the metals at temperatures characteristic of flue gases, and subsequent adsorption of the metals onto fine particles. The normal boiling points for zinc and cadmium (907 and 767 degrees centigrade respectively), for example, are substantially lower than normal combustion chamber temperatures. Moreover, many metals form volatile metallic chloride compounds, and some metals may be dispersed at the molecular level in combustible materials (e.g. additives in plastics).

Most toxic metals emitted from municipal incinerators are associated with fly ash particles due to the high surface area of those particles. Fly ash particles may be removed from the gas stream with fabric filters or electrostatic precipitators (Neal and Scubel 1987). At temperatures between 125 and 140 degrees centigrade, removal efficiencies greater than 99 percent have been demonstrated for all heavy metals except mercury. Eighty to 90 percent removal efficiencies have been demonstrated for mercury at temperatures lower than 140 degrees centigrade (EPA 1987). Nevertheless, the EPA projects emissions of approximately 20,000 kg/yr of cadmium, 2550 kg/yr of chromium, 1,260,000 kg/yr of lead, and 148,000 kg/yr of mercury (EPA 1987).

Greennburg, et al. concluded, "If all urban refuse were burned in incinerators, the levels of some toxic elements in urban air would probably become intolerable." However, they also noted that neither of the two facilities used during their tests recovered energy, and steam generation could reduce flue gas temperatures, thereby decreasing emissions of volatile metallic compounds. Furthermore, neither facility was equipped with the best available control technology. Thus, their results might overstate the potential human health risks (Greenburg, et al. 1978:572).

Greenburg, et al also concurred with Campbell (1976) and Law, et al. (1978), who suggested that separation of noncombustible and heavy combustible materials from light combustible materials might substantially reduce emissions of toxic metals. Lead, for example, is primarily concentrated in the noncombustible fraction of refuse, and cadmium is found primarily in heavy gauge plastics. Thus, emissions of toxic metals from RDF facilities may be substantially lower than emissions from mass burn facilities.

In general, EPA modelling studies indicate that metallic emissions do not pose a substantial human health risk, relative to dioxins and furans. For existing facilities, 0.5 cancer deaths per year are predicted. For new facilities, 0.4 deaths are predicted. Results of the EPA study also indicate that mercury and lead standards for direct

inhalation should not be exceeded for new and existing facilities. However, there is some concern about exposure of children to lead and mercury deposited in soils surrounding combustion facilities (EPA 1987). Moreover, removal of metals adsorbed onto fly ash particles only transfers the threat of contamination to another medium.

Municipal Waste Combustion Ash

In addition to the health risks from incinerator stack emissions, there are still unresolved questions about incinerator ash. Greenburg, et al (1978) reported relatively high concentrations of lead, cadmium, zinc, copper, manganese, and mercury in fly ash from municipal waste combustion facilities. Diaz, et al. (1982) stated that concentrations of cadmium, cobalt, lead, and zinc could exceed 1000 ppm. Given these high concentrations and the acidic characteristics of municipal leachate, some heavy metal leaching is probable.

However, in a series of studies conducted by Versar, Inc. and NUS Corporation for the EPA, only lead leached from fly ash at levels deemed hazardous, under laboratory conditions. When subjected to the EP Toxicity Test, the Toxicity Characteristic Leaching Procedure (TCLP), and the Monofilled Waste Extraction Procedure, the concentration of lead in one ash sample exceeded the maximum concentration of 5 mg/l. A second ash sample also failed the TCLP test for lead (Darcey, "Incinerator Ash Dilemma", 1988). Although the EPA study did not detect high levels of lead or other heavy metals in leachates from ash disposal sites (Darcey, "Incinerator Ash Dilemma",1988), several research groups have reported heavy metals in leachate samples from ash monofills (EPA, Subtitle D Report, 1988).

Sawell, et al (1987) and Bridle, et al. (1987) have conducted leachate studies that may simulate the behavior of ash in landfills more realistically than the batch extraction tests used by the EPA. In their procedure, called the Sequential Chemical Extraction procedure, the ash sample is divided into five fractions: exchangeable metal ions, surface oxide and carbonate-bound metal ions, Fe-Mn oxide bound metal ions, organically-bound metal ions and sulfides, and residual metal ions. Using this procedure, Sawell, et al. investigated the behavior of metals in a lime scrubber residue. They concluded that significant proportions of cadmium, copper, lead, zinc, boron, and cobalt may leach under normal landfill conditions. Using the same procedure, Bridle, et al. concluded that 80 percent of the cadmium, 40 percent of the zinc, and significant fractions of the copper, cobalt, and nickel in MSW fly ash is readily leachable. They suggest that MSW fly ash should not be co-disposed with MSW.

Sawell, et al also investigated the leaching properties of polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, polychlorinated biphenyls, polyaromatic hydrocarbons, chlorinated benzenes, and chlorinated phenols. They concluded that organic contaminants in MWC ash are not mobile under normal landfill conditions. Results of the EPA studies support that conclusion. The EPA results indicate that negligible concentrations of PCB's, dioxins, furans, and semivolatiles are present in leachates from landfills where ash has been deposited. According to the EPA study, dioxins are relatively immobile in the natural environment. However, the EPA did not consider its data on dioxins to be complete, and it recommended further evaluations of dioxins and furans in leachates from co-disposal sites and monofills (Darcey, "Incinerator Ash Dilemma",1988).

Data on groundwater contamination by municipal waste combustion ash is insufficient to support a final conclusion. The EPA report cautioned that data collected during its study "are not necessarily representative of all municipal waste combustion facilities or ash fills." (Darcey, "Incinerator Ash Dilemma," 1988).

Available data does indicate that municipal waste combustion facilities may pose substantial human health risks unless such facilities are properly designed and operated. Human helath risks from exposure to dioxins and furans may be overestimated and, consequently, the overall health risks due to incinerator emissions may be overstated. However, health risks associated with heavy metals in incinerator ash may be underestimated.

Accurate risk assessment may be academic. Empirical data on health risks associated with dioxins will probably not quell the public's fears of those chemicals nor lessen public opposition. Recent events in California presage such opposition. California's solid waste planners envisioned 50 percent incineration, but their plans were curtailed by public opposition. In Los Angeles, public opposition forced the mayor to stop planning for the controversial LANCER incinerator proposal. In San Francisco, the city was forced to withdraw from commitments for refuse incineration, and in San Diego, the SANDER incinerator project was abandoned (Smith 1987). Like opposition to landfills, the ultimate impact of public opposition toward incineration on the decision making process at the local level is difficult to assess.

Incineration can be appealing to local decision makers because, in addition to reducing the volume of waste that must be landfilled, incinerators may generate revenue. The 1978 Public Utility Regulatory Policies Act guarantees a market for the energy produced by municipal incinerators (Pollock 1987). Currently, 106 of the 140 operational municipal waste combustion facilities produce energy;

1,251.55 megawatts of electricity is being generated. If all planned facilities are completed, 3,565.18 megawatts will be generated from combustion of refuse (EPA 1987).

In the Subtitle D census, nine states including Mississippi, which reported no capacity problems, cited resource recovery as an important component of current or future solid waste disposal plans. The EPA has estimated that 350 municipal waste combustion facilities with a total capacity of 250,000 tons per day will be operational by the year 2000 (EPA, Subtitle D Report, 1988).

The increase in the total capacity of municipal waste combustion facilities coupled with the shortage of landfill capacity in some areas of the nation foreshadows a gradual decline in the number of landfills in the United States. That trend is substantiated by data on the number of new landfills permitted in recent years. In 1981, 559 new landfills were approved by the states. In 1982, 524 landfills were approved, and in 1983, only 416 landfills were approved (EPA, Subtitle D Report, 1988). Even if incineration and recycling are maximized, however, some landfill disposal capacity will be necessary. Incineration and materials recovery or recycling are only waste reduction and/or stabilization processes. Furthermore, as indicated in Table 1, municipal refuse only constitutes about 55 percent of the solid wastes currently deposited in sanitary landfills (Stearns and Woodyard 1977).

"Refuse" has never been well defined. The American Public Works Association (1970) lists all items in Table 1 plus abandoned vehicles and sewage sludge as refuse. Rimberg (1975) uses a definition similar to the definition of Stearns and Woodyard. In this terminology, "refuse" shall define wastes normally collected in packer trucks and acceptable at incineration or composting facilities (i.e. household, commercial, small wood and lumber, garden, and street and park wastes), approximately 55 percent by weight of the total waste stream deposited at sanitary landfills.

Weight is obviously not equivalent to volume, and volume is the parameter of importance in landfill disposal. Unit weights and compaction factors for the various components of the municipal waste stream were not available, but data presented by Stearns and Woodyards indicate that the in situ densities after compaction of the various components only varies by approximately 100 lb/cu. yd. Therefore, the data in Table 1 has been used to appoximate volume reductions.

Table 1: Composition of Total Solid Waste Stream

Type of Waste		Percent by Weight
Household		
Paper and Paperboard	35.6	
Glass	8.4	
Metal	8.9	
Plastic	7.3	
Rubber and Leather	2.8	
Textile	2.0	
Wood	4.1	
Food Waste	8.9	
Yard Waste	20.1	
Miscellaneous Inorganic	1.8	
Total	100.0	31.3
Commercial		18.6
Solid Fill		33.4
Small Wood and Lumber		1.2
Large Wood and Lumber		1.1
Demolition		5.9
Garden		2.7
Street and Park		2.6
Industrial		1.7
Bulky Items		0.6
Rubber Tires		0.7
Agricultural		0.2

Sources: Stearns and Woodyard, Subtitle D Report

Efficacy of Incineration

In areas where landfill space is limited, incineration may offer significant benefits for local solid waste managers. By the 1990's, EPA estimates that 25 percent of the municipal refuse generated in the United States will be incinerated (EPA 1987). In some other countries, the percentage of refuse incinerated is already much greater than 25 percent. For example, Sweden incinerates 51 percent of its refuse; Japan incinerates 64 percent, and Switzerland incinerates 75 percent (Pollock 1987). Thus, refuse incineration has substantial potential for growth in the United States.

However, local decision makers must balance the benefits of volume reduction and energy recovery against potential problems associated with incinerators. Despite recent advances in refuse combustion technologies, a significant portion of any municipal waste stream must still be landfilled after the combustible portion of the waste has

been incinerated. Furthermore, many incinerators still suffer from mechanical shutdowns that force rerouting of solid wastes to landfills, and air pollution problems may force a facility to close permanently.

Incineration of refuse in a mass burn facility generally reduces the volume of waste by 70 to 90 percent. Since refuse constitutes approximately 55 percent of the wastes buried at sanitary landfills, actual volume reductions are approximately 40 to 50 percent. Some facilities have claimed much greater volume reductions. For example, the Dade County Resource Recovery Plant, a 2500 tpd facility, claims to bury only 3 percent by volume of its incoming refuse.

However, the Dade County facility buries demolition debris, which constitutes approximately 6 percent of the waste stream, and "processed materials", probably white goods, which constitute approximately 0.6 percent of the waste stream, in a nearby sanitary landfill. Furthermore, the facility recovers aluminum, approximately 1 percent of typical refuse; ferrous metals, approximately 8 percent; and glass, approximately 10 percent, from the waste stream (Neal and Schubel 1987). Thus, the actual volume reduction by incineration was only 80 percent of incoming wastes, and a minimum of 36 percent of the municipal waste stream, industrial wastes, tires, agricultural waste, and solid fill, was not taken into account.

Refuse derived fuel facilities burn approximately 40 to 60 percent of incoming refuse (Diaz, et al 1982), and the volume of ash residues is approximately 10 to 20 percent of the volume of the refuse derived fuel (General Electric 1975). Thus, volume reduction for incoming refuse is approximately 50 to 60 percent, and reduction of the total waste stream is approximately 30 percent. For example, the defunct City of St. Louis RDF facility, which accepted only packer truck wastes, no bulky items, produced 72 tons of ferrous metal, 632 tons of fuel, 296 tons of landfill debris, and 10 tons of ash for every 1000 tons of incoming refuse (General Electric 1975).

Although refuse derived fuel facilities have less potential for volume reduction than mass burn plants, such facilities may offer substantial benefits in terms of reliability and marketing, important considerations for local solid waste managers. Refuse incinerators must reliably process a continuous stream of wastes to maintain community support. Furthermore, facilities that sell energy to repay construction and operation costs must supply a dependable source of electricity or steam to their customers.

Some mass burn facilities have excellent records for availability. For example, in five years of operation, the three furnace mass burn plant in Pittsfield, Massachusetts

never sent garbage to landfill unscheduled and never operated the auxillary burners to maintain boiler output to the steam customer (Reason 1986). However, average availability of mass burn facilities is only about 80 percent (Diaz, et al 1982). The frequency of downtimes necessitates duplication of equipment and/or an alternative landfill disposal site. In addition, furnaces must be capable of firing fossil fuels to provide energy when refuse is unavailable. Limitations on availability of mass burn facilites primarily result from clogging of grates with partially burned or incombustible materials in the fuel. Other potential problems include corrosion due to low redox conditions, slagging, fouling, and temperature excursions.

Corrosion, caused by production of carbon monoxide and hydrogen sulfide under low redox conditions, may be a direct result of poor distribution and stratification of air or fuel or both, a result of the heterogeneity of refuse. Carbon monoxide and hydrogen sulfide may react to remove the protective coating on furnace tubes. Corrosion may also occur when hydrogen chloride and sulfur oxides in flue gases contact surfaces with temperatures below the dew point of water and dissolve to form corrosive acids (Diaz, et al 1982). In addition, slagging may cause corrosion. Slag is molten ash that hardens and forms deposits on furnace walls. In addition to corrosion, slagging interferes with heat transfer. (Diaz, et al 1982). Fouling, which occurs when the temperature of the flue gases at the superheater tubes is higher than the ash fusion temperature, may cause erosion of the tubes. Tubes may also be damaged by temperature excursions. At the Dade County facility, superheater tubes had to be replaced every 2000 hours. Other major components that may be damaged by temperature excursions include the grate mechanism and the air pollution control equipment (Diaz, et al 1982). All these problems relate indirectly to the heterogeneous nature of municipal refuse and the concomitant problems with temperature control. To prevent temperature excursions and low redox conditions, refuse must be properly mixed; the grate must be evenly loaded and operated at the correct speed, and the combustion and overfire air flows must be correctly adjusted (Reason 1986). Alternatively, solid waste managers may opt to process refuse into a more manageable fuel.

Firing refuse derived fuels rather than raw refuse may ameliorate many potential problems associated with refuse incineration. Since RDF contains less incombustible material, grates become clogged less frequently. Also, the heating value of the fuel is more uniform, so temperature excursions decrease, and consequently, less complicated combustion control systems are necessary. If glass is recovered during preparation of RDF, slagging may be reduced, and RDF can be distributed more uniformly, so low redox conditions may be prevented. In addition, RDF offers better boiler response; RDF preparation may remove some

pollutants at the fuel preparation stage, and refuse derived fuels have lower ash content and higher heat content than raw refuse (Reason 1986). The heat content of refuse derived fuels is approximately 7000 BTU/lb for RDF versus approximately 4500 BTU/lb for mixed refuse (Salimando 1988, Wolpin 1988)

Furnaces designed to fire RDF have greater thermal efficiency than mass burn furnaces because less excess air is necessary to achieve proper mixing; less heat is lost with incombustible materials, and the fuel contains less moisture. Therefore, RDF plants may produce 8 to 15 percent more energy per ton of refuse than mass burn plants (Remirez 1986). Furthermore, RDF can be burned with other fossil fuels, and consequently, municipalities may be able to market RDF more easily than they can market steam or electricity.

Local solid waste managers have several choices with regard to energy marketing: steam generation, electrical generation, or RDF production. Steam generation produces 0.65 to 0.7 BTU for every BTU of energy input; whereas, generation of electricity produces only 0.2 BTU per BTU input. However, steam generation is limited by proximity to an industrial user (Remirez 1986). Consequently, local decision makers are often forced to choose on-site electrical generation (Diaz, et al 1982). Approximately 74 percent of new facilities produce electricity only (Darcey,"Resource Recovery Market Update," 1988).

This decision necessitates construction of a power plant and installation of transmission and distribution equipment. If the community is forced to assume the responsibilities of an electrical utility, refuse disposal may become subservient to power generation. Some localities have regulations limiting recycling to maintain fuel supplies for power plants (Pollock 1987). The alternative to on-site power generation is production of refuse derived fuels. Refuse derived fuels can be transported relatively easily, and such fuels can be fired in conventional boilers with ordinary fossil fuels.

However, refuse derived fuels have not been completely accepted by industry for co-firing in coal utility boilers due to poor experiences in a number of locations including plants in Milwaukee, Chicago, and Rochester, all closed (Eldred 1987). Moreover, the heating value of coal is twice the heating value of RDF; the moisture content of RDF is twice the moisture content of coal, and the ash content of RDF is three times greater than coal. Practical consequences of the differences between coal and RDF are greater storage and materials handling problems (Diaz, et al 1982). New RDF facilities generally prepare RDF for firing by itself in specially designed boilers (Eldred 1987).

Crude RDF preparation, shredding and ferrous metals removal, may cause long boiler downtimes due to feeder screw wear, jams from glass and metal, combustion control problems, and low ash fusion temperature due to glass content of the fuel. Generally, however, preparation of refuse derived fuels transfers reliability problems from the boiler to waste processing equipment. For example, the City of Lakeland reported no problems with the boiler in their RDF facility, but avaiability of the preparation plant was only 55 percent (Reason 1986).

Preparation plants at modern RDF facilities usually include trommeling, size reduction, magnetic separation, air classification, and screening. Size reduction, particularly, has occasioned many problems in the past. Normally, a high horsepower hammermill is used to shred the refuse. High maintenance costs are incurred by wear on the shredder. Sand and glass are driven into the combustible material, and some items, such as compressed gas cylinders, may explode (Reason 1986). Furthermore, the process is energy intensive. Greater than 50 percent of the energy requirements for production of RDF are associated with size reduction (Diaz, et al 1982).

Problems associated with shredding may be alleviated if refuse is processed through a low horsepower flail shredder and a trommel screen prior to size reduction. The Madison, Wisconsin RDF facility achieved 94 percent availability with that system (Reason 1986). Past experience indicates that the Madison plant may be exceptional; however, its success and the success of the Pittsfield plant demonstrate that both RDF and mass burn facilities may effectively reduce the volume of refuse if such facilities are well designed and well operated.

Ultimately, the choice between mass burn and RDF may be based upon economics. For projects of 750 tons/day or less, mass burn is more economical. For larger projects, RDF becomes more cost effective. Available RDF processing equipment becomes cost-effective at high capacity. Because smaller combustion units are feasible for RDF plants, the cost of processing equipment is offset by savings on combustion equipment. Downstream boiler accesories are also smaller, and requirements for combustion air are lower. Moreover, costs for operation of RDF plants are lower due to their higher thermal efficiency (Remirez 1986).

Modular incineration facilities are cost-effective for treatment of small volumes of refuse. Such facilities use starved air furnaces in contrast to excess air conditions typical of furnaces at mass burn plants. Because modular incinerators are small, use of such facilities for refuse processing may ameliorate many institutional problems associated with transporting large quantities of refuse to one location. NIMBYists may also be appeased by reductions

of truck traffic, etc. However, experience indicates that starved air furnaces are more susceptible to damage from temperature excursions than waterwall furnaces used in mass burn facilities, and therefore, their useful lifetimes may be shorter (Reason 1986).

Currently, 47 percent of operational refuse incinerators use mass burn technology, 34 percent are modular incinerators, and 18 percent are RDF incinerators. Seventy percent of incinerators being planned will be mass burn facilities, 16 percent will be modular plants, and 14 percent will be RDF plants (Darcey, "Resource Recovery Market Update," 1988). Since the average size of facilities being planned is 742 tons/day, the popularity of mass burn plants probably reflects the cost effectiveness of mass burn technology for low volume facilities.

In addition to cost effectiveness, local decision-makers must weigh human health risks, public opposition, plant availability, requirements for final disposal, and, particularly, uncertainty associated with air pollution control when incineration is being considered as a refuse treatment process. Currently, the greatest obstacle to further growth of incineration as a waste treatment process is uncertainty about air pollution regulations. Failure of the grate or fuel processing system may close a plant for weeks, but failure to meet air pollution control requirements may force a plant to close permanently. Fears of local decision-makers are reflected in the volatility of the resource recovery market. In 1987, 20,585 tons/day of total new capacity was ordered. However, 35,655 tons/day was cancelled (Darcey,"Resource Recovery Market Update,"1988). New air pollution regulations for refuse incinerators are scheduled for promulgation in September, 1991 (EPA 1987).

Decisions about incineration may also be influenced by public opinion on resource recovery. Incineration recovers a portion of the energy stored in the organic fraction of refuse, and therefore, it may generate favorable public opinion. However, incineration, particularly mass burn, may destroy potentially valuable recyclable materials.

Efficacy of Recycling

Recycling offers local decision-makers an opportunity to reduce the volume of waste that must be landfilled or incinerated while conserving resources and, possibly, generating some revenue. It is certainly appealing from the viewpoint of conservation ethics. The wastage of potentially valuable resources is probably the most odious aspect of landfill disposal. Currently, 50.1 million tons of paper and

paperboard, 11.8 million tons of glass, 12.6 million tons of metal, 10.3 million tons of plastic, and 40.8 million tons of organic materials are buried annually (EPA, Subtitle D Report, 1988).

Recycling not only conserves resources and reduces the volume of the solid waste stream, however, it also saves energy and reduces pollution. Production of recycled paper requires 23 to 70 percent less energy than production of paper from virgin materials. It also reduces air pollution by 74 percent, water pollution by 35 percent, and water consumption by 58 percent. Production of aluminum cans from scrap rather than bauxite cuts energy consumption by 95 percent, air pollution by 95 percent, and water pollution by 97 percent. Glass recycling cuts energy consumption by 4 to 22 percent, air pollution by 20 percent, and water consumption by 50 percent (Walsh and O'Leary 1988). Production of fabricated plastic products from scrap cuts energy consumption by 85 to 90 percent, including the energy value of the petroleum feedstocks (Pollock 1987, Bernheisel 1988). Thus, additional recycling of post-consumer materials could increase the efficiency of resource utilization nationwide, allowing further economic expansion.

Recycling of municipal solid waste is limited, however, by irremediable contamination of recyclable materials in mixed refuse, the efficiency of materials separation technology, public participation in source separation programs, and the availability of markets for secondary materials. Recycling can probably only achieve, realistically, a 15 to 30 percent reduction in the municipal refuse stream (Eldred 1987; Waste Age June, 1988; Alter 1987). Thus, the maximum volume reduction of the total waste stream will only be about 85 percent.

Many materials in refuse, such as oily rags, paper mixed with food residues, diapers and old paint cans, can not be reused due to contamination (Waste Age June 1988). Other materials can not be effectively separated from mixed refuse. For example, a squeezable ketchup bottle is constructed from six layers of plastics; each layer with different characteristics and different functions. (Pollock 1987). These different plastics can not be separated with available technology, and the quality of products manufactured from undifferentiated plastics is generally lower than the quality of products manufactured from virgin resins. In addition, the recyclability of some refuse constituents may be limited by the efficiency of available refuse processing equipment.

Materials Recovery

Due to the heterogeneity of municipal refuse, a variety of unit processes are necessary for maximum materials recovery. Size reduction, air classification, magnetic separation, and screening are most common. Other processes include eddy current separation, froth flotation, and optical sorting. Size reduction is generally the first unit process in refuse treatment schemes. Size reduction or shredding is necessary because many components of refuse are constructed from disparate materials. For example, the common "tin can" has steel in its body, a tin liner, a paper wrapper, and, possibly, an aluminum top (Diaz, et al 1982). To separate these materials into discrete pieces of steel, paper, and aluminum, shredding would be necessary. In addition, shredding decreases the average particle size and homogenizes the waste. Consequently, the size of downstream processing equipment can be reduced, and mechanical failures in downstreanm equipment are lessened. Currently, hammermills are the most common shredders being used.

After being shredded, the pieces of the tin can would be segregated by density. A variety of density separation equipment is available: air classifiers, vibrating tables, mineral jigs, dense media separators, water elutriation separators, centrifuges, and cyclones. However, air classifiers are most common (Hecht 1983, Savage, et al 1986). In an air classifier, the lightweight pieces of paper from the tin can's wrapper would flow upward in a column of air to be recovered with other low density materials such as cloth and plastic. By weight, the light fraction from air classification generally consists of approximately 60 percent paper, 6 percent plastic, and 34 percent miscellaneous. Because air classifiers actually separate materials by their aerodynamic drag/weight ratio, however, some lightweight aluminum (e.g. aluminum foil) might be segregated into the light fraction. Fifty percent of aluminum in mixed refuse may be lost in the light fraction (Diaz, et al 1982).

At this stage of the separation process, the lightweight fraction from the air classifier would be processed through a trommel, a rotating cylindrical screen, to remove inorganic fines. Screening efficiency of trommels is approximately 70 to 90 percent (Hecht 1983). Products of the trommel screen would probably be burned as refuse derived fuel. However, some processes, particularly, the Cal Recovery Systems (CRS) process, can recover paper fibers from screened lights. In the CRS process, screened lights are pulped to produce a slurry; the slurry is screened to remove plastics, then centrifuged twice to remove other contaminants and, finally, dewatered (Diaz, et al 1982).

The heavy fraction of the waste stream, including the steel and aluminum pieces of the tin can, glass, rocks, wood, and other high density materials, would exit the bottom of the air classifier. By weight, the heavy fraction from air classification consists of approximately 24 percent ferrous metals, 36 percent glass, 2 percent aluminum, 4 percent paper, 2 percent plastic, and 32 percent miscellaneous (Diaz,et al 1982). To further separate these materials, the heavy fraction would be processed through a magnetic separator.

Approximately 85 to 90 percent of the ferrous metals in the heavy fraction of the waste stream could be extracted with a magnetic separator. However, the ferrous scrap might be contaminated with 0.2 to 1 percent by weight of paper, plastic, and rags. Ferrous materials might be further separated into a heavy fraction for sale in the scrap iron market and a light fraction for sale in the higher priced detinning market. The light ferrous fraction would be treated with hot sodium hydroxide to convert the tin liner from the can into sodium stannate. The sodium stannate would subsequently be converted into elemental tin by electrolysis. Approximately 6 lb. of tin could be recovered from every ton of tin cans processed by this technique. The detinned steel might be used for steel production or precipitation of copper from low-grade ores and mining wastes.

Aluminum from the top of the tin can might be recovered from the nonferrous heavy fraction by eddy current separation, dense media separation, vertical symmetric field separation, the Black-Clawson method or ALCOA's Air Knife method. Eddy current separators can acheive separation efficiencies ranging from 70 percent at a feed rate of approximately 40 tons/day to 99 percent at a rate of 2 tons/day. However, 20 to 25 percent of the product from eddy current separation may be organics trapped or entrained by aluminum particles. Eddy current separators were being used at the Milwaukee Americology resource recovery plant and the New Orleans resource recovery plant. However, both plants have been shutdown (Eldred 1987; Diaz, et al 1982). The Black-Clawson method is apparently being used for recovery of aluminum at the Dade County resource recovery facility (Neal and Schubel 1987). The Dutch States Mines process, a dense media separator, is a commercially available. However, it is generally too expensive for recovery of aluminum from municipal solid wastes. The Air Knife and vertical symmetric field separators are only available as prototypes (Diaz, et al 1982). None of the processes have been proven economically feasible. Wilmington, Delaware and Dade County, Florida apparently have operational modules for mechanical recovery of aluminum, but units in Ames, Iowa; Monroe County, New York; New Orleans, Louisiana; and Milwaukee, Wisconsin are not operational (Neal and Schubel 1987).

Problems include contamination by organics, low throughput rates necessary to achieve satisfactory separation, and inadequate separation of aluminum from other nonferrous metals (e.g. Cu and Zn).

Glass might also be recovered from the heavy nonferrous fraction by froth flotation or optical sorting. Wilmington, Delaware aparently uses froth flotation for recovery of glass (Neal and Schubel 1987). New Orleans and Monroe County also had froth flotation units, but those units are not operational (Eldred 1987). Optical sorting is not economically feasible for full-scale operation (Diaz, et al 1982).

Greater separation efficiency can be achieved when mixed refuse is hand-sorted to remove recyclable materials. However, this method is labor intensive and costly. Generally, therefore, only materials with high resale values such as aluminum, steel, and, possibly, glass are recovered. Cardboard, newsprint, and beverage containers made from polyethylene terephthalate (PET) might also be recycled, but the number of different materials that can be recovered at such facilities is limited. One sorting facility in Marin County, California recovers wood, yard wastes, cardboard, glass bottles, aluminum cans, scrap metal, paper, rags, PET bottles, and white goods. Overall, the facility recycles 26 percent of central Marin County's refuse. However, a recycling surcharge of one dollar per dwelling unit per month is necessary to maintain the facility (Mattheis 1988).

Most taxpayer-consumers would not willingly pay a surcharge to support a recycling program. Although 70 percent of adult Americans believe recycling can solve much of the country's solid waste disposal problem, a recent survey showed that only 15 percent would willingly pay a tax or fee to support a program that did not break even (Waste Age June 1988).

Source Separation

In addition to being costly, sorting facilities recover materials contaminated by contact with mixed refuse. These materials have less value than clean post-consumer materials. Source separation alleviates both cost and cleanliness problems. However, such programs depend upon public participation. In a study on the feasibility of source separation programs, 90 to 95 percent of the householders who were polled indicated that they would participate in such programs. However, active response has seldom exceeded 5 percent (Vesilind and Pierce 1982). One exception is the city of Santa Monica, California. That city claims an overall participation rate of 35 percent, and a participation rate of 80 percent in an area of single family dwellings (Darcey,"Recycling Interest Surges,"1988). Two

states, New Jersey and Rhode Island, now mandate source separation (Waste Age June 1988). Hundreds of communities across the nation have programs. Nevertheless, Harvey Alter of the U.S. Chamber of Commerce has noted, "...the number of source separation programs appears to be growing much faster than the mass of material recovered and sold."(Alter 1987:31)

Market Demand

Collection and separation of materials is not the principal problem of recyclers. Finding markets for the recovered materials is the principal problem. In the words of Deborah Levin of the Institute of Scrap Recycling Industries,"You cannot force the reuse of recyclables, collection and removal of items from the waste stream does not mean they will be recycled." (Eldred 1987:64). Collection of recyclable items that can not be used does not benefit the community, and it is not cost efficient. Curbside collection of recyclable materials costs $20 to $40/ton (Pollock 1987), and it may be more expensive if participation rates are low. If the recovered materials can not be used, then those materials must be landfilled at an average cost of approximately $20/ton. In 1985, two-thirds of the plastic soft drink containers collected under New York's mandatory deposit law had to be buried due to poor scrap markets (Pollock 1987).

Markets for recycled materials can be difficult to find for a variety of reasons. Often, virgin raw materials are cheaper than post-consumer materials. This advantage of virgin over recycled materials is sometimes reinforced by tax policies, such as depletion allowances for mining and oil, and long distance freight rates (Waste Age June 1988) In addition, recycled materials can not compete with the quality of virgin materials in many markets. Referring to recycled paper, Basil Snyder, president of Garden State Paper, a newsprint recycler in New Jersey, stated, "...there are limits as to how much paper the industry can accept because low cost recycling technology to produce the kind of high grade paper that markets demand does not exist." (Eldred 1987:64)

The limitations of recycled paper became evident when the state of Illinois tested the feasibility of using paper with 50 percent recycled content in state offices. The study showed that the recycled paper worked well in one and two sided printing. However, reading the recycled paper printed on both sides was difficult. The recycled paper also tested well in small copiers, but jammed in high speed copiers. The study concluded that the specifications for the content of recycled material in the paper should be lowered from 50 percent to 40 percent (Waste Age May 1988).

Uses of recycled paper are also limited by regulations of the Food and Drug Administration. FDA Food Additive Regulation 21 CFR S176.260 prohibits the use of paper for food, drug, or cosmetic containers if such paper had previously been used for shipping or handling poisonous or deleterious substances. Since it would be impractical to determine prior uses of paper extracted from mixed refuse, the practical consequence of that regulation is the preclusion of all paper from mixed refuse for the manufacture of food or cosmetic containers (Diaz, et al 1982).

Currently 28 percent of the waste paper in the United States is recycled. More than half of the recycled paper is used to produce paperboard; 25 percent is used for tissue, newsprint and similar purposes (Waste Age June 1988). Eventually, the fibers become too short to be recycled for paperboard or newsprint, but short fibers can still be used to produce tissue, insulation, and animal bedding (Pollock 1987). Recycled paper is also used in the manufacture of molded packaging, for example, egg cartons, and cushioning materials for packing boxes (Waste Age June 1988). When the fibers become too short or contaminated to be useful for other purposes, the paper can be used in compost or refuse derived fuel. Alter of the U.S. Chamber of Commerce has noted that the market for recycled paper is limited by the availability of commercial and industrial sources of high grade secondary paper (Alter 1987). However, substantial overseas markets for recycled paper are available. In 1986, over four million tons of recycled paper were exported (Pollock 1987).

Further growth in the use of recycled paper depends upon the effectiveness of source separation programs. Centralized sorting operations are too costly, particularly recovery of mixed paper, which must be hand separated into computer printouts, newsprint, magazine paper, and office paper. In addition, contamination lowers the value of the paper recovered at sorting facilities. There is little commercial interest in mechanical recovery of paper fiber; problems of quality and consistency are too great (Eldred 1987). Given these constraints, the maximum recovery rate for paper is probably about 50 percent (Pollock 1987).

Already, over 50 percent of the aluminum cans used in the United States are being recovered (Eldred 1987). By the 1990's, manufacturers would like to increase the rate of recovery to 75 percent (Waste Age June 1988). Aluminum is outstanding among recyclable materials because, unlike most post-consumer materials in municipal refuse, its value on the open market is greater than its cost-avoidance value (Pollock 1987), the amount of money saved from landfill fees, etc. when a recycled material is removed from the solid waste stream. The price paid for recycled aluminum is approximately $500/ton (Waste Age June 1988). Due to its

high resale value, professional scavengers have begun to collect aluminum cans. These scavengers are considered to be largely responsible for the success of aluminum recycling efforts (Alter 1987). Despite their success, the impact of aluminum can recycling efforts is minimal in terms of municipal solid waste volume reduction. Aluminum cans comprise only about 0.5 percent of the municipal solid waste stream. Overall, the recovery rate for aluminum is only about 30 percent. A 50 percent recovery rate is perhaps feasible. (Pollock 1987).

Glass is recovered at a lower rate than aluminum because the price paid for cullet, crushed glass, is lower and the glass recovery transportation networks are less developed (Eldred 1987). Furthermore, glass must generally be sorted by color to receive the highest resale price. Currently, only 10 to 12 percent of the glass in municipal refuse is recycled (Eldred 1987; Waste Age June 1988). Recycled glass is used in the production of approximately 25 percent of the bottles manufactured in the United States (Darcey,"Recycling Interest Surges," 1988). Post-consumer glass can also be used in the manufacture of fiberglass, asphalt aggregate, bricks, and glass polymer composite for sewer pipes (Waste Age June 1988; Diaz, et al 1982).

Demand for clean, color-sorted cullet is relatively strong within transportation range of glass manufacturing facilities since, for every ton of cullet used, 1.2 tons of raw materials are saved. Furthermore, an increase of 10 percent of the amount of cullet introduced into glass furnaces saves 2 to 5 percent on energy consumption (Pollock 1987). Two-thirds of the bottles discarded by the consumer market could probably be recycled by the glass industry (Eldred 1987). However, due to the small number of markets for recycled glass and the difficulty of color-sorting it, 50 percent recovery is more likely (Pollock 1987).

Plastic is the most difficult material to recycle in municipal refuse. Currently, plastics comprise 4.1 percent of the solid waste stream including 11 percent of all beverage containers (Bernheisel 1988). By the year 2000, the plastics segment of the waste stream should increase to approximately 10 percent (Eldred 1987). However, at present, only about one percent of the plastic used in the United States is recycled (Waste Age June 1988). Major problems of plastic recycling include the inability to visually distinguish one plastic resin from another (Pollock 1987), rigid specifications for recovered materials, lack of processing equipment, transportation limitations (Bernheisel 1988), and, most important, the slow growth of markets for recycled plastics (Eldred 1987).

The major market for recycled plastic is fiberfill for sleeping bags, jackets, and automobile seats. Twenty to twenty-five percent of the soft drink containers made from

polyethylene terephthalate (PET) are recovered, primarily for production of fiberfill (Pollock 1987, Alter 1987, Waste Age June 1988). In addition to fiberfill, recycled PET is used in continuous filament yarn, unsaturated polester molding resin for bathtubs and shower stalls, rigid urethane foam insulation for refrigerators, thermoformed sheets for audiocassette cases, and strapping for large packages. Currently, recycled PET accounts for 85 percent of the plastic recovered in the United States. High density polyethylene from milk bottles, bleach bottles, and motor oil containers is also recovered (Bernheisel 1988), and mixed plastic wastes have been used to manufacture plastic lumber (Pollock 1987). Despite these applications for recycled plastics, however, plastic has generally been characterized as the bane of the recycling industry. Joseph Garbarino of the Marin Resource Recovery Center has said, "The market prices aren't high enough for most waste plastics to cover the cost of recovering it. So, most of it ends up in the landfill." (Mattheis 1988)

Basil Snyder provided the most succinct summary of the current role of recycling when he stated," We cannot recycle our way out of the garbage crisis." (Eldred 1987) Stearns and Woodyard (1977) suggested that a well designed mechanical recovery system with modules for RDF, ferrous metals, nonferrous metals, and glass could recover 90 percent of the municipal refuse stream. Thus, landfill disposal would be necessary for only 55 percent of total solid wastes. However, the economic feasibility of this alternative is dubious. Sorting facilities and source separation programs might recover 40 percent of municipal refuse. Eighty percent of solid wastes would be landfilled under that scenario.

A greater number of recycling programs could increase the amount of material recovered from municipal refuse in the near future. However, the efficacy of recycling as a disposal option varies across the nation. Ed Klein, head of the EPA's Municipal Solid Waste Task Force noted, "The solid waste situation is not the same in Texas as it is in Massachusetts. Recycling is going to be more expensive in a rural setting, where the stops are further apart, than it might be in a suburb." (Salimando 1988:52).

Even if all recycled materials could be readily transported to markets, the markets might not be able to absorb the materials. Already, there are fears that public collection programs could cause a glut of recycled materials (Alter 1987, Eldred 1987). Ultimately, increased demand for recycled materials depends upon the development of new markets. However, development of new markets will require changes in the way products are designed, changes in the quality of products expected by the public, or artificial equalization of the price difference between recycled and virgin raw materials (Eldred 1987). EPA Assistant

Administrator for Solid Waste, J. Winston Porter has concluded, "...people who believe that we can recycle 80 or 90 percent of the solid waste stream are crazy. We'll always have a need for landfills and incinerators." (Darcey,"States Take Action," 1988:56)

Disposal Options

Options for disposal of municipal solid wastes other than direct landfill disposal, incineration and recycling are limited. Some possibilities are ocean disposal, composting, pyrolysis, and anaerobic digestion.

Ocean Disposal

Currently, no municipalities in the United States dump refuse into the ocean. In the past, ocean disposal of refuse has fouled beaches with floating trash, killed birds and fish that ingested plastic particles, and caused significant economic losses for fishermen whose ship's screws and water intakes became fouled with floating debris. The 1972 Ocean Dumping Act and the 1974 International Dumping at Sea Act effectively banned ocean disposal of raw refuse. However, the feasibility of ocean disposal of baled refuse and incinerator ash has been investigated.

Neal and Schubel (1987) reported the results of a series of studies if ocean disposal of baled refuse. During that study, adverse effects on marine life were identified; furthermore, some marine organisms colonized the bales. Toxic compounds apparently diffused to ambient levels within short distances of the bales or precipitated as metallic sulfides within the bales. No data were reported, however, concerning pathogenic contamination of filter feeders in the vicinity of refuse bales. During a year long study, refuse bales maintained their integrity, however, methane and other gases trapped in bales caused some bales to become buoyant (Neal and Schubel 1987).

Incinerator ash dumped into the ocean caused heavy metal toxicity in menhaden, lobster larvae, and sea scallops. Substantial mortality also occurred from direct burial of benthic organisms and ingestion of ash particles. Researchers speculated that ocean disposal of unstabilized ash could destroy the benthic population in disposal areas.

Ash stabilized in concrete blocks might be used to create artificial reefs or construction areas. Some research has been conducted with blocks constructed from various combinations of coal fly ash and scrubber sludge. These blocks maintained their compressive strength for five years, and no adverse effects on marine life were observed. Moreover, no significant differences were observed between

marine life on the reef constructed of experimental blocks and a control reef. However, coal ash generally has lower levels of toxic metals than incinerator ash, so these results may not be applicable to disposal of incinerator residues.

Neal and Schubel suggest that the ocean does possess some capacity to accept certain types of wastes without unreasonable degradation, and they note that the ocean is one of only three alternatives for final disposal of solid wastes: land, ocean, or atmosphere. However, more research is necessary to determine the adverse effects of ocean disposal, and currently, it is not a viable alternative to landfill disposal.

Composting

Composting is a viable method for land disposal of municipal refuse, and it provides some benefits to the land as a soil additive. Compost improves the texture, aeration, and water holding capacity of soils (Diaz, et al 1982). It also provides all micronutrients necessary for plant growth, improves the availability of phosphorus, increases the cation exchange capacity of the soil, and stimulates microbial activity (Diaz, et al 1982; Ambrose 1983). Other advantages of composting include the following: (1) capability to treat other wastes such as septage, wastewater treatment sludge, or food industry wastes; (2) marketable end product; (3) 40 to 70 percent volume reduction; (4) stabilization of pathogens and some toxic components of municipal solid wastes; and (5) compatibility with recycling and RDF production (Goldstein 1987).

In conjunction with RDF production and/or recycling, composting can reduce the volume of refuse by approximately 75 percent (Goldstein 1987). Thus, only about 60 percent of the waste stream must be landfilled. In the past, however, most compost has been used for landfill cover because markets for compost have been limited (Goldstein 1988). Use of compost for landfill cover can increase landfill disposal capacity by 20 to 30 percent because space normally occupied by daily cover is available for waste disposal. However, no data are available on the effectiveness of compost, a humus-like material, for fire protection, a major function of daily cover.

Aerobic, thermophilic microbial processes prevalent in most composting operations can kill most pathogens in refuse and may also destroy potentially hazardous aromatic and aliphatic compounds. Aromatic compounds, particularly, would degrade slowly under the anaerobic conditions prevalent in landfills. In addition, compost may adsorb refractory organics and toxic metals in refuse, thereby, retarding their migration to groundwater. Therefore, some waste

professionals advocate composting of refuse prior to landfilling as a technique for groundwater protection. However, the cost of composting will probably limit such applications.

Composting costs range from $25 to $60/ton, depending upon landfill fees, financing methods, preprocessing equipment costs, and compost markets (Goldstein 1987). Substantial preprocessing is necessary for preparation of high-quality compost from unsegregated municipal refuse. Unless composting is combined with materials recovery or RDF production to permit preprocessing costs to be prorated (Goldstein 1988), composting may not be economically competitive with other disposal options. Recovery of glass, plastics, and metals also increases the quality of the compost, and high quality compost is necessary for sales to the general public, nurseries, and landscapers.

Compost has not been widely used as a soil additive in the United States; therefore, markets are limited. At application rates of approximately 20 tons/acre, compost can improve crop yields, but it is not a good fertilizer. The N,P,K content is only 0.5 to 1.6 percent nitrogen, 0.2 to 0.7 percent phosphorus, and 0.2 to 1.2 percent potassium (Diaz, et al 1982; Ambrose 1983). Compost may increase the availability of phosphorous in the soil by stimulating microbial mineralization, and in addition, it may prevent leaching of nitrogen. However, the carbon:nitrogen ratio of the compost must be 20:1 or less to prevent microbial immobilization of nitrogen. The C:N ratio of unsegregated municipal refuse in the United States is frequently 55:1 or higher. To lower the ratio, paper products can be separated from the waste stream, or the refuse can be co-composted with sewage sludge (Diaz, et al 1982).

Agricultural interests in the United States have not shown an interest in large-scale application of MSW compost despite its beneficial effects as a soil additive. The lack of interest may be due primarily to lack of information about compost, particularly its water holding capacity. However, farmers with knowledge of compost have also rejected it because the costs of transportation and application have often exceeded profits from increased crop yields. In addition, the optimum time for compost application, early spring, is also the worst time for heavy equipment, necessary for compost application, to work fields due to problems with compaction of wet soils (Diaz, et al 1982).

Despite the high cation exchange capacity of compost, some potential for heavy metal leaching and plant uptake exists when compost is used as a soil additive. Dyer and Razvi (1987) concluded that plant uptake and leaching could be controlled if the total amount of compost applied to soils was regulated. If controlled application is necessary,

however, sales to the general public seem questionable. Unless agricultural interests can be stimulated, reclamation of drastically disturbed lands, such as strip mines, is probably the best use for MSW compost (Diaz, et al 1982).

Although scarcity of markets and cost may limit future growth of composting, the number of composting facilities in the United States will probably increase substantially in the near future. Currently, six facilities compost municipal refuse: New Castle County, Delaware; Portage, Wisconsin; Fillmore County, Minnesota; St. Cloud, Minnesota; Sumter County, Florida; and Dodge County, Kansas. Further growth of composting as a treatment process has been limited by uncertainty among local decision-makers due to limited experience and expertise with composting. Nevertheless, thirty-four communities have plans for composting facilities (Goldstein 1988), and expertise and experience with composting has increased lately due to interest in sludge composting.

Composting may become the preferred disposal option for small, rural communities. Such communities may not be able to afford landfill disposal under the new Subtitle D regulations; transportation to regional landfills may also be too expensive, and incineration will not be cost-effective for low throughput volumes. Furthermore, haul distances may be short enough for farmers that a sustainable market can be developed for the compost produced by such communities.

Pyrolysis

Pyrolysis converts the organic fraction of municipal refuse into fuel oil or fuel gas by combusting the refuse at temperatures ranging from 900 to 3000 degrees fahrenheit in the absence of oxygen. Some pyrolysis processes accept only the organic fraction of refuse and require front-end waste processing systems. Other processes accept all components of refuse and produce a slag of metals, glass, and other inorganic materials in addition to carbon char and fuel oil or fuel gas produced by pyrolytic conversion of the organic fraction. Final products of pyrolysis are a function of temperature. For example, the Garrett process can produce fuel gas at temperatures of 1400 degrees fahrenheit or fuel oil at 900 degrees fahrenheit (General Electric 1975).

Nonslagging processes, such as the Garrett process, generally recover ferrous metals, approximately 7 percent by weight of incoming refuse, and glass, approximately 8 percent, during processing. The carbon char produced during pyrolysis, equivalent to approximately 5 percent by weight of the incoming refuse, is comparable to low grade coal, but generally, it is not marketable. Approximately 7 percent by weight of the incoming refuse, not including the carbon

char, is landfilled. For every 1000 tons of refuse processed, 228 tons of oil, comparable to No. 6 fuel oil, are produced (General Electric 1975).

Slagging processes produce an inert slag, equivalent to approximately 20 to 25 percent by weight of the incoming refuse, in addition to fuel oil or fuel gas. Such processes may also produce some carbon char, depending upon the temperature of the pyrolysis process. Union Carbide's PUROX process, for example, produces 220 pounds of slag and 25 million std. cubic feet of fuel gas for every 1000 tons of refuse. Temperatures during the Union Carbide process are 2600 to 3000 degrees fahrenheit (General Electric 1975).

Pyrolysis reduces the weight of refuse by 80 to 90 percent. Volume reduction of the total waste stream is approximately 45 percent (Stearns and Woodyard 1977). Despite the high volume reduction potential of pyrolysis, growth of pyrolysis as a refuse treatment process is unlikely. The process is expensive; capital and operating costs are approximately $55/ton of refuse (Diaz, et al 1982). Furthermore, air and water pollution problems are probable (General Electric 1975). Four pyrolysis facilities have been built in the United States. None is currently operational (Neal and Schubel 1987).

Anaerobic Digestion

Anaerobic digestion of methane generates biogas, which contains 60 to 70 percent methane, and a dark, tarry slurry. Approximately 60 to 70 percent of volatile solids in refuse are destroyed during anaerobic digestion. Thus, volume reduction of the refuse is approximately 45 percent, and volume reduction of the total waste stream is approximately 25 percent. The methane in biogas can be sold after scrubbing to remove carbon dioxide, hydrogen sulfide, hydrogen, and ammonia. Approximately 185 cubic meters of methane are produced per ton of refuse. The slurry can be easily dewatered and land-farmed; however, supernatant from the slurry has high biochemical oxygen demand. In addition to potential water pollution problems, cost is a major disadvantage. Anaerobic digestion costs approximately $42/ton of refuse (Diaz, et al 1982).

The Pompano Beach, Florida pilot plant is the only facility in the United States that digests refuse anaerobically. It was designed to treat 100 tons/day, but operational difficulties caused by design deficiencies in the front end processing equipment (i.e. shredder, electromagnet, and air classifier) have reduced the plant's capacity (Diaz, et al 1982). This method is probably not a viable method for refuse management unless hydraulic refuse conveyance systems are developed.

Miscellaneous

A variety of miscellaneous processes for treatment of refuse have been researched including enzymatic hydrolysis, acid hydrolysis, cellulose embrittlement, and hydrogenation. The first three processes treat only the cellulosic fraction of refuse (50 to 65 percent). Both enzymatic and acid hydrolysis convert cellulose to glucose for production of ethanol and single-cell protein (Diaz, et al 1982). In acid hydrolysis, the organic fraction of refuse is slurried and treated with mineral acid at temperatures of 450 to 500 degrees fahrenheit (Hecht 1983). In enzymatic hydrolysis, cellulase, produced by fungi, is used to break the cellulose bonds. Cellulose embrittlement converts cellulose to a friable powder through chlorine, nitric acid, and hydrochloric acid treatment in combination with low levels of heat. In the hydrogenation process, hydrogen reacts with the organic fraction of refuse under elevated temperatures and pressures to produce synthetic gaseous liquid fuels (Hecht 1983). None of these processes is a viable alternative for refuse management.

The City of Philadelphia has developed another alternative. The city has plans to build a 485 ton/day processing plant to convert garbage into a cellulosic fiber for use in agriculture, construction, or paper production, and an inert, granular vermiculite-like soil aerator through size reduction, ferrous metal recovery, drying, stabilization, and sanitation with ozone (Remirez 1986). The success of the plant has not been reported. The cellulosic fiber could be burned as refuse derived fuel. Large-scale markets for the soil aerator are probably limited.

RDF production with materials recovery is probably the best available technology for refuse management in urban areas, and composting with materials recovery is probably the best available technology in rural areas. Composting with materials recovery could reduce landfill requirements by approximately 45 percent. RDF production with recovery of metals and glass could also reduce landfill capacity by 45 percent (Stearns and Woodyard 1977). If those technologies could be maximized, total landfill capacity requirements would be approximately 186 million cubic yards/year by the year 2000, assuming an *in situ* waste density of 1000 lb/cubic yard. Current capacity requirements are approximately 225 million cubic yards/year. Thus, landfill capacity requirements could be substantially reduced.

However, rapid growth of composting and RDF production is doubtful. Growth of composting will be slow due to the hesitancy of local decision-makers to accept unproven

technology and the lack of markets for compost. Public opposition, technological problems, and economies of scale that favor mass burn technology will slow the growth of RDF production. Moreover, both techniques are expensive relative to landfill disposal.

Future of Landfill Disposal

Landfill disposal will continue to be the predominant method of refuse management for the foreseeable future. Currently, it is the only viable option for final disposal. To protect human health and the environment, therefore, the solid waste community must identify appropriate landfill technology for municipal refuse management.

The EPA addressed this issue when it published proposed new regulations for municipal landfills (Federal Register 53 (168): 33314-33422). The new regulations would promulgate a performance standard for municipal landfills, which, ideally, would limit the health risk from exposure to groundwater contaminated by MSW leachate. However, those regulations were premised upon effective leachate control, achieved through utilization of low permeability liners and covers.

Recent evidence indicates that clay and/or synthetic liners, which represent the best available control technology for municipal landfills, may not prevent leakage of some components of MSW leachate. In the absence of effective containment, costly and possibly ineffective corrective action would be required under the proposed regulations. In addition, low permeability covers, which minimize leachate generation, prevent rapid biological stabilization of municipal refuse, prolonging the liability of landfill owners and operators and delaying return of the land to productive use.

Given the continued importance of landfill disposal in municipal solid waste management, a re-evaluation of landfill technologies is warranted. The effectiveness of clay and synthetic liners for groundwater protection at municipal landfills will be reviewed. Innovative leachate treatment technologies will also be reviewed.

Adequacy of Current Technology

The level of technology employed at most municipal landfills in the United States is not much advanced beyond the technology of the ancient Greeks, who invented the city dump. Generally, "out of sight, out of mind" has been the philosophy of landfill disposal, yet as early as 1932, there were indications that the "out of sight, out of mind" philosophy was fallacious. In 1932, Calvert reported

increases in the concentrations of iron, total hardness, total solids, carbon dioxide, and total organic nitrogen in well water near a pit, used for disposal of liqour drained from cooked garbage. Although this evidence suggested that buried refuse could adversely affect groundwater resources, the potential health risks of landfill disposal were largely ignored by government and academia until the 1950's.

The first significant studies of the leaching properties of municipal wastes were conducted, during the 1950's, by the State of California Water Pollution Control Board and the University of California (Zanoni 1972). For many years, the conclusions of the California reports embodied the prevalent thought on groundwater contamination at municipal landfills:

1) A sanitary landfill, if so located that no portion of it intercepts groundwater, will not cause impairment of the groundwater for either domestic or irrigational uses.

2) A sanitary landfill, if so located as to be in intermittent or continuous contact with groundwater, will cause the groundwater in the vicinity of the landfill to become grossly polluted and unfit for domestic or irrigational uses.

These maxims became the basis for the "dilute and disperse" principle of landfill disposal. That philosophy was premised upon the attenuation capacity of the vadose zone and the dilutional capacity of groundwater flow systems.

During the late 1960's and 1970's, numerous incidents of well contamination, attributed to leachates from municipal landfills, were reported (Walker 1969; Coe 1970; Apgar 1975; Shuster, Peoples Avenue Case Study, Fox Valley Case Study, Sayville Case Study, 1976; DeWalle and Chian 1979). In general, findings from investigations of those incidents substantiated the conclusions of the California reports. At 14 of 20 sites where leachate migration was investigated, refuse had been deposited below the seasonal high water table. Nevertheless, many researchers were prompted to further investigate the characteristics of municipal leachates and the movement and attenuation of leachate plumes.

Results of those investigations indicated that attenuation capacity was limited in some hydrogeologic environments. In 1969, Cartwright and Sherman noted that limestone quarries and gravel pits were usually not acceptable refuse disposal sites (Zanoni 1972). In 1970, Schneider suggested that clay and silt environments had substantial attenuation potential, but sand and gravel, sand and silt, fractured rock, and limestone were poor attenuators (Garland and Mosher 1975). In 1971, Zanoni

noted that natural attenuation mechanisms were limited in unconsolidated materials and limestone or shale with fissure, faults, or fractures. Thus, good attenuation was limted to areas characterized by uniform, extensive deposits of clay, sandy clay, and clayey silt.

Where natural attenuation mechanisms have been insufficient for protection of groundwater, clay and/or synthetic liners have been constructed to contain leachate. However, recent evidence indicates that many municipal landfills in the United States have been improperly located and/or designed. Prior to 1980, 5 percent of the municipal landfills built in the United States were located in karst terrain or below the seasonal high water table; approximately 5 percent were located in wetlands, and 10 percent in floodplains. Forty-four percent of the landfills had no liner system; 50 percent had no surface water run-off/run-on control system, and only 11 percent had leachate collection systems (EPA, Subtitle D Report, 1988).

Despite the evidence of potential health risks, no federal regulations specifically addressed disposal of municipal refuse until 1979. The Criteria for Classification of Solid Waste Disposal Facilities and Practices (40 CFR 257), promulgated in 1979 under the authority of sections 1008 and 4004 of the Resource Conservation and Recovery Act, banned refuse disposal in "open dumps". The criteria mandated daily cover to control disease vectors, fire, and birds; protected endangered species; prohibited open burning; regulated landfills in floodplains; required control of explosive gases; restricted public access; outlawed the discharge of pollutants in violation of the Clean Water Act, and limited the discharge of pollutants to underground drinking water sources in excess of the maximum contaminant levels of the Safe Drinking Water Act.

Facilities that complied with the provisions of the 1979 criteria were termed sanitary landfills. Those facilities that did not satisfy any of the criteria were termed open dumps. Under the provisions of the Guidelines for Development and Implementation of State Solid Waste Management Plans (40 CFR 256), open dumps were supposed to be closed or upgraded. However, the EPA was not given the authority to enforce the part 257 regulations. Therefore, little improvement in the design and operation of landfills was attained.

Among landfills built after 1980, there was no significant improvement in the percentage of facilities located in floodplains, karst terrain, wetlands or below the seasonal high water table. Landfills sited in floodplains increased from 10 percent for facilities built prior to 1980

to 14 percent for facilities built after 1980. The percentage of landfills located in karst terrain or below the seasonal high water table remained the same. The percentage built in wetlands decreased slightly (EPA, Subtitle D Report, 1988).

Use of engineering design controls did increase slightly after promulgation of the part 257 criteria. Landfills with leachate collection systems increased from 11 percent to about 18 percent. Landfills with surface water run-on/run-off control systems increased from 50 percent to 75 percent, and landfills with natural or synthetic liner systems increased fron 56 percent to 75 percent (EPA, Subtitle D Report, 1988).

Frequency of Leakage

However, improvements in engineering design apparently did not occasion a decline in the incidence of leakage from municipal landfills. In 1976, Shuster had concluded:

> Information on leachate migration from damage cases and specific site studies coupled with general information on disposal site locations and operations in the United States indicate that at least one-fourth and possibly as many as three-fourths of the municipal land disposal sites in the United States have leachate migration problems (Shuster, Peoples Avenus Case Study, 1976).

Data from the 1984 Subtitle D census indicates that approximately 25 percent of existing municipal landfills still leak. In the census, groundwater contamination violations were reported at approximately 6 percent of all municipal landfills, yet only about 25 percent of municipal landfills monitored groundwater in 1984. Thus, violations must have been detected at approximately 24 percent of the facilities that did monitor groundwater (EPA, Subtitle D Report, 1988).

Due to inadequate monitoring, the actual extent of groundwater contamination can not be determined (EPA, Subtitle D Report, 1988). However, a prediction greater than 25 percent does not seem unreasonable, assuming that those facilities with proper monitoring systems were generally better designed and operated than the majority of municipal landfills. Thus, in the absence of effective regulations to prevent leakage, future incidents of well contamination are probable.

Available information for landfills, where leakage has been documented, is insufficient to support conclusions about the effectiveness of clay or synthetic liners and covers. Most landfills where leachate contamination occurred probably did not employ impermeable covers and liners. Prior

to 1980, 44 percent of municipal landfills had no liner system, and only 1 percent had synthetic liners (EPA, Subtitle D Report, 1988). The evidence is also not sufficient, nonetheless, to preclude the possibility of leakage. Monitoring at municipal landfills has been inadequate, and the actual extent of groundwater contamination can not be determined. Only 25 percent of the landfills in the Subtitle D Census monitored groundwater.

Effectiveness of Clay and Synthetic Liners

However, Johnson (1986) observed that 90 percent of caps, clay and synthetic, exhibited impacts from biological and physical forces that would impair the function of the landfill cap. Problems included erosion, freeze-thaw, soil shrinkage, waste subsidence, internal gas pressure, and floating objects, such as tires.

Daniels and Brown (1987) noted that average field permeability for clay liners was 1/100,000 rather than 1/10,000,000 cm/s, the standard permeability for clay liners. Factors that contributed to lower permeabilities included: (1) use of liners without a sufficient number of lifts, (2) compaction at relatively low moisture content, (3) use of relatively thick lifts of soil, (4) use of soils deficient in clay, (5) use of relatively light compaction equipment, (6) poor construction quality assurance, (7) dessication, (8) freezing, and (9) interactions between clay liner and leachate. Concentrated organic acids may cause dessication of clay liners. However, the concentrations in municipal leachate are generally too low to cause problems. Clay liners were banned in 1982 for construction of hazardous waste landfills because the liners did not meet the agency's zero leakage, zero penetration goals. Problems with synthetic liners may be caused by puncture holes, chemical attack, gas pressure, or improper joint sealing. At 27 lined facilities, 10 failures were reported that caused releases of pollutants (Daniels and Brown 1987).

Moreover, significant theoretical evidence exists to indicate that clays and synthetic liner materials may not retard the migration of some components of municipal leachate, particularly trace organic compounds. Griffin and Shimp (1978) observed that organic materials, measured as COD, were poorly attenuated by clay minerals. Fuller (1978) reached a similar conclusion when he studied the interaction between organics, measured as TOC, and eleven different soils. Although soils with high concentrations of organic matter can adsorb organics, such soils are not typically found at the depth of most landfills. Haxo (1988) indicates that synthetic liners may not retard the migration of some components of municipal leachate. Haxo demonstrated that some aromatic and chlorinated aliphatic hydrocarbons in dilute aqueous solutions could permeate through synthetic

liner materials within 24 hours. However, he did not investigate the permeation of trace organics in municipal leachate. His earlier research on permeation of gases and water vapor in municipal leachate through synthetic liners is also insufficient to support general conclusions about permeation rates of organic species in leachate (Haxo,et al 1985)

The preamble to the proposed rulemaking states that "...even the best liner and leachate collection system will ultimately fail due to natural deterioration..." Given the inevitable failure of liner systems, remedial action will often be necessary to achieve the performance standard promulgated by the new regulations.

The regulatory impact analysis (RIA) indicates that, under the proposed rule, the annualized cost of corrective action for landfills with 10 meter points of compliance would be 97.8 million dollars. For landfills with 150 meter points of compliance, the annualized cost would be 129.4 million dollars. However, the RIA also indicates that the costs of corrective action have been underestimated:

> For the proposal, we assumed that all model scenarios that met the design goal with the appropriate design in place would not trigger corrective action... we recognize that there are situations in which landfills will not be properly designed and the performance standard will not be met, triggering corrective action. Accounting for these types of situations is, however, beyond the scope of this analysis.

The case histories of landfills, where remedial cleanup activities have been undertaken, also encourage some skepticism about the estimated costs of corrective action in the regulatory impact analysis. For example, the Subtitle D Report to Congress mentioned a remedial cleanup action at one 600 acre unlined municipal landfill that cost tens of millions of dollars, not including the cost of replacing two municipal wells with a combined capacity of 120 million gallons per day. Using these costs for cleanup at a single landfill as an indicator, the estimated costs in the RIA for remedial activities at 6000 landfills nationwide seem relatively low.

In addition to its cost, moreover, corrective action presents numerous other potential problems including intermedia contaminant transfer, control of exposures to residual contamination, management of wastes generated by remedial activities, and control of the sources of releases. The preamble to the proposed rulemaking admits that source control could be difficult or impossible : "...treatment or capping to prevent further leaching may not be technically feasible or completely effective in eliminating further releases above health based contaminant levels." If leachate

generation and migration can not be controlled, then landfill management could degenerate into a continuous pump and treat process.

Leachate treatment is probably a better strategy for achieving the performance standard than leachate minimization and containment. Leachate treatment would not only avoid costly and/or ineffective remedial cleanup, it could also increase the rate of stabilization of the waste, allowing the land to be returned to productive use and minimizing the liability of landfill owners and operators.

Conclusion

A variety of leachate treatment systems have been developed, but a permeable attenuation zone underneath the landfill, combined with a leachate collection system and secondary leachate treatment, might be the least expensive and most manageable. In situ treatment would be less expensive than on-site aerobic treatment or off-site treatment at a sewage treatment plant, and it would not produce large quantities of potentially hazardous sludge. In situ treatment would also require less maintenance and supervision than on-site anaerobic treatment.

In situ leachate treatment technology would supplement natural attenuation mechanisms (i.e. anaerobic biodegradation within the refuse mass and sorption by in situ soils) at municipal landfills where such mechanisms do not provide sufficient groundwater protection to achieve the performance standard. Although in situ soils might be sufficient protection against groundwater contamination at some landfills, an engineered attenuation zone would probably be desirable at many landfills. Hydrogeological conditions are often complex, and accurate modeling of leachate attenuation and migration can be difficult. Consequently, predicting the capacity of in situ soils to to attenuate pollutants at a proposed landfill site could be difficult, and if initial predictions were proven erroneous, costly corrective action might be required. In contrast, the properties of an engineered attenuation zone (e.g. cation exchange capacity, organic carbon content, etc.) would be known, and the quality of the construction could be controlled. An engineered attenuation zone might no always perform to design standards due to cracking, channelling, etc. However, a landfill with an engineered zone would be more likely to achieve the performance standard without corrective action than a landfill with an attenuation zone consisting entirely of in situ soils.

In situ treatment at municipal landfills would be analogous to land treatment of hazardous sludges. However, management of in situ treatment processes at municipal landfills would be more difficult than management of land

treatment facilities; the attenuation zone of the landfill would be difficult or impossible to repair or redesign after initiation of waste disposal operations. Therefore, leachate collection systems would probably be necessary as a management tool at landfills located in areas of high precipitation. The leachate collection system would function as a primary leak detection system. If the collected leachate contained high concentrations of biodegradable organic matter after leaching through the attenuation zone, then it could be recirculated. Collected leachate might also receive additional treatment at an on-site spray irrigation or aerobic lagoon facility to remove compounds not amenable to anaerobic degradation (e.g. aromatic compounds). Such simple aerobic systems would not require skilled personnel; furthermore, such systems would not require large amounts of energy for aeration because the bulk of the organic matter in the leachate would be removed by anaerobic processes within the refuse.

The design of an engineered attenuation zone would depend upon the properties of the waste to be deposited at the landfill, the natural attenuation capacity of the landfill site, average annual precipitation at the site, and groundwater depth. However, a typical zone might include an attenuation layer, a high permeability drainage layer, and a low permeability layer to increase the efficiency of the leachate collection. The attenuation layer might include a layer of limestone to increase the precipitation of metallic ions and a layer of organic material, for example, coal or peat, to retard the migration of organic pollutants. The low permeability layer might be a clay or synthetic liner.

In situ leachate treatment technology would supplement natural attenuation mechanisms (i.e. anaerobic biodegradation within the refuse mass and sorption by in situ soils) at municipal landfills where such mechanisms do not provide sufficient groundwater protection to achieve the performance standard.

Recommendations

(1) Data generated by groundwater monitoring requirements of the proposed regulations should be collected and disseminated by the EPA with information about landfill age, hydrogeological characteristics, refuse characteristics, and landfill design and operation. This data is necessary to assess the effectiveness of landfill liner systems.

(2) Research on interactions between flexible membrane liners and trace organics in municipal leachate is necessary.

(3) Research on interactions between clay liners and trace organics in municipal leachate is necessary.

(4) Composting might be appropriate for refuse treatment on tribal lands for the following reasons: (1) Refuse on Indian reservations probably has high organic content. (2) Some tribal lands probably need humus. (3) Labor should be available cheaply for hand-sorting glass, aluminum, etc. (4) Tribal leaders may have authority to spread compost on tribal lands (landowners do not have to be convinced to try compost). (5) Composting operation might create jobs in economically depressed areas. (6) Composting plants on tribal lands could demonstrate benefits to the general agricultural community. Composting should be encouraged by the EPA.

(5) EPA should encourage RDF production with materials recovery.

(6) EPA should encourage recycling at all levels: (1) nationwide recycling curriculum, (2) grants for municipalities to develop recycling programs, (3) encourage phaseout of depletion allowances for mining, (4) encourage realistic timber prices for lumber companies in national forests, (5) encourage tax allowances for recycling equipment, (6) set standards for recycled content of government purchases, (7) legislation for plastic resin codes, etc.

(7) Research on stimulation of sulfate reducing bacteria in municipal landfills with waste gypsum. Most metals in municipal refuse precipitate as metallic sulfides. Since co-disposal of incinerator ash with municipal refuse is probable in the absence of regulations barring the practice, addition of excess sulfide should ensure precipitaion of most metals. Sulfate-reducing bacteria might generate sulfide from waste gypsum under conditions prevalent in municipal landfills. Potential problems are low pH, hydrogen sulfide generation, and excess hardness.

Future Research

Because groundwater protection standards manadated by the new Subtitle D regulations are premised upon effective leachate control through utilization of low permeability liners and covers, further research is necessary to quantitate the migration potential of trace organic constituents. Research by Haxo, Fuller, and Griffin and Shimp suggests that clay and/or synthetic liners and in situ soils may not provide sufficient groundwater protection. However, their research does not provide sufficient data for modeling migration of trace organic constituents in municipal solid waste leachate. Fuller and Griffin and Shimp

researched interactions between soils and gross organic parameters, but they did not investigate attenuation of specific organic constituents. Haxo investigated interactions between synthetic liners and trace organics found in municipal leachates, but his work does not address multicomponent leachate interactions.

Future research must focus upon determination of the parameters necessary for effective modelling of the migration of trace organics in municipal solid waste leachates. Bench-scale studies can be used to determine the solubility parameters of various organics in municipal leachates and the partition coefficients between organic constituents and various soils. These parameters could be input to existing models (e.g. Subtitle D Risk Model) to predict migration potentials. Column studies, using an apparatus similar to Haxo's equipment, could generate calibration data for the migration models with various combinations of synthetic liners and soils. Thermal chromatography, developed by Ruska Laboratories in Houston, could be used to determine concentrations of trace organics in the various solid phases. In addition, toxicity studies could be used to estimate the risk associated with various leachates after passage through different liner combinations. Although such studies would not address macroscopic problems, such as cracking and punctures, quantitative data to estimate the effectiveness of low permeability liners would be generated. Such data could be used to determine whether further research on alternative in situ treatment systems (i.e. permeable liners) is warranted.

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